Air Transport and Deposition of Actinides at the Rocky Flats Environmental Technology Site

FY99 Report

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EXECUTIVE SUMMARY

The Rocky Flats Environmental Technology Site (RFETS, or Site) has been a source of airborne actinides throughout its history. Over time, small amounts of plutonium (Pu), americium (Am), and other actinides have been deposited on or mixed with surface soils at the Site. Wind or mechanical disturbance of the contaminated soil can result in actinide-laden soil particles becoming airborne. These resuspended particles will be transported some distance downwind before being deposited on the ground or in water by a variety of mechanisms that remove particles from the air, such as rainout or dry deposition. As a result, airborne migration is one of several transport pathways that redistribute actinides in the environment in the vicinity of the Site (other primary pathways include soil erosion and surface and groundwater movement).

Fiscal year 1999 (FY99) air pathway studies have focused on emission of actinides into the air from contaminated soils or debris (resuspension), transport of airborne actinides from source areas at the Site to other parts of the Site or off Site (dispersion), and removal of actinide-contaminated particles from the air to soil or water surfaces (deposition). Sensitivity and comparison studies were also performed to evaluate dispersion model performance and to focus future work.

Emission Estimation

Actinide resuspension due to natural phenomena at the Site is episodic in nature and influenced primarily by meteorological variables (wind speed and rainfall); particle and soil properties (moisture level and particle density); and surface characteristics (density and type of vegetative growth, and snow cover). Given the density of vegetation within the contaminated soil areas on Site, the primary source of contaminated soil resuspension is considered to be the dust-laden vegetation and litter, with little potential for direct resuspension from exposed soil surfaces (assuming no disturbance of the area).

Past wind tunnel experiments on Site relate dust resuspension to ambient wind speed and currently provide the best method for estimating emissions. Site wind tunnel data indicate dust resuspension varies with wind speed raised to the third power.

An equation was derived relating hourly particulate and actinide emissions to wind speed, underlying surface-soil contamination levels, and the presence or absence of snow cover. This equation was used to calculate hourly emissions for five actinides due to natural resuspension mechanisms for the 1996 calendar year. The calculated emissions were used as input to dispersion and deposition simulations.

In addition, calculation methods were identified for a variety of anthropogenic emission mechanisms, such as excavation, traffic, maintenance of storage piles, etc. These methods will be used in conjunction with the natural resuspension equation identified above to calculate actinide emissions from specific remediation or decontamination and decommissioning (D&D) scenarios in future work.

Dispersion and Deposition Modeling

In parallel with the emission estimation activities, a model was developed that can be used to simulate dispersion and deposition of actinides for a variety of emission events or scenarios. An annual scenario was modeled representing the "chronic" resuspension of actinides. Airborne actinide concentrations due to these ongoing emissions were estimated at a variety of locations on and around the Site. The airborne concentrations were also converted to dose units using U.S. Environmental Protection Agency (EPA) conversion factors. Annual deposition of actinides that have become airborne due to chronic resuspension mechanisms also was estimated at locations in and around the Site.

Maximum actinide concentrations due to natural resuspension mechanisms were predicted to occur along the Site's eastern fenceline. This location was anticipated, given the predominant westerly winds at the Site. Similarly, the annual predicted Pu-239 and Am-241 deposition contours were found to extend toward the east-southeast from the eastern edge of the Industrial Area. The patterns of annual deposition for the uranium isotopes were variable because of the differing locations of the sources (areas of higher surface soil concentrations).

Comparison and Sensitivity Analyses

Air sampling data for Pu-239 were available for comparison with model results. Model predicted concentrations were found to be higher, by one-to-two orders of magnitude, than the measured concentrations. It is expected that the overprediction is partially due to dilution of the resuspendable dust attached to vegetation surfaces, relative to the actinide density in the underlying surface soils. Other potential factors were also identified. These factors will be taken into account in future modeling to decrease potential overprediction.

Three sensitivity analyses were also performed to examine: 1) the inclusion of an additional source at the background or fallout level of Pu-239, 2) the performance of a general resuspension factor previously developed for the Site, and 3) the effect of plume depletion on predicted concentrations. The first analysis showed that inclusion of an additional source at background levels would not substantially increase predicted actinide concentrations. The second analysis showed that the Site-wide resuspension factor developed previously produces results that match measured actinide concentrations fairly well (within the same order of magnitude). This is not surprising because the resuspension factor was developed from on-Site sampling data collected just to the east of the 903 Pad. However, the general resuspension factor can only be used to calculate annual average actinide values, whereas the method developed in this study can be used to vary emissions and impacts on an hourly basis.

The third analysis showed that removing the mass of particulate that is deposited to ground or surface waters from the plume would decrease predicted air concentrations (and dose) by 20 to 26 percent. The deposited particulate fraction, which was "double counted" in the study reported here, should be taken into account in future modeling.

Information was also presented on a related study that examined the strength of the correlation between meteorological variables and measured actinide air concentrations on Site. Measured Pu-239 concentrations to the east of the 903 Pad were shown to be strongly correlated with the occurrence of strong, westerly winds (as expected). The amount of precipitation, on the other hand, did not directly correlate with measured concentrations.

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Abbreviations and Acronyms

ADOM	Acid Deposition and Oxidant Model
Am	Americium
AME Group	Actinide Migration Evaluation Group
ARAC	Atmospheric Release Advisory Capability
BLM	U.S. Bureau of Land Management
CAPARS	Computer Assisted Protective Action Recommendation System
CAP88-PC	Clean Air Act Assessment Package-1988 (Version 1.0)
CFR	Code of Federal Regulations
Ci	Curie(s)
cm	Centimeter(s)
cm^3	Cubic centimeter(s)
Cs	Cesium
D&D	Deactivation and decommissioning
DEM	Digital Elevation Model
DOE	U.S. Department Of Energy
EPA	U.S. Environmental Protection Agency
FY99	Fiscal year 1999
g	Gram(s)
hr	Hour(s)
ISC3	Industrial Source Complex model, Version 3
ISCLT	Industrial Source Complex model, Long-Term
ISCST	Industrial Source Complex model, Short-Term
kg	Kilogram(s)
km	Kilometer(s)
lb	Pound(s)
m	Meter(s)
m^{-1}	Inverse meter(s)
m^2	Square meter(s)
m^3	Cubic meters(s)
mg	Milligram(s)
Mg	Megagram(s)
mm	Millimeter(s)
mph	Mile(s) per hour
MPRM	Meteorological Processor for Regulatory Models
mrem	Millirem
MRI	Midwest Research Institute
m/s	Meter(s) per second
NRC	Nuclear Regulatory Commission
NREL	National Renewable Energy Laboratory
NTIS	National Technical Information Service
NWS	National Weather Service
OAQPS	Office of Air Quality Planning and Standards

Abbreviations and Acronyms (continued)

OU3	Operable Unit 3
pCi	Picocurie(s)
$PM_{2.5}$	Particulate matter < 2.5 micrometers
PM_{10}	Particulate matter < 10 micrometers
Pu	Plutonium
RAAMP	Radioactive Ambient Air Monitoring Program
RARC	Regional Atmospheric Response Center
RESRAD	Residual radioactive material model
RFETS	Rocky Flats Environmental Technology Site
RMRS	Rocky Mountain Remediation Services, L.L.C.
s, sec	Second(s)s
s^{-1}	Inverse second(s)
Site	Rocky Flats Environmental Technology Site
T-3/T-4	Trench 3/Trench 4
TRAC	Terrain-Responsive Atmospheric Code
U	Uranium
USFS	U.S. Forest Service
USGS	U.S. Geological Survey
UTM	Universal Transverse Mercator
µm	Micrometer(s)
VKT	Vehicle kilometers traveled
VMT	Vehicle miles traveled
yr	Year

1.0 INTRODUCTION

The Rocky Flats Environmental Technology Site (RFETS, or Site) has been a source of airborne actinides throughout its history. Over time, small amounts of plutonium (Pu), americium (Am), and other actinides have been deposited on or mixed with surface soils at the Site. Wind or mechanical disturbance of the contaminated soil can result in actinide-laden soil particles becoming airborne. These resuspended particles will be transported some distance downwind before being deposited on the ground or in water by a variety of mechanisms that remove particles from the air, such as rainout or dry deposition. As a result, airborne migration is one of several transport pathways that redistribute actinides in the environment in the vicinity of the Site (other primary pathways include soil erosion and surface and groundwater movement).

The Actinide Migration Studies Group (currently called "AME Group"), convened by the U.S. Department of Energy (DOE) in 1996, recommended that a conceptual model for actinide transport be developed to establish relationships between the Site's physical and chemical characteristics and the fate and transport of radionuclide contaminants in the environment. This report documents efforts during fiscal year 1999 (FY99) to improve estimates of airborne actinide migration and deposition in the conceptual model, prepare a modeling tool to use in evaluating various emission scenarios, and provide preliminary air pathway concentration and dose estimates.

1.1 Background

The major source areas that contribute to airborne actinides at the Site are the 903 Pad and the adjacent "lip" area. The 903 Pad was contaminated with Pu- and Am-laden cutting oil stored in metal drums, which over time leaked onto the soil beneath the drums. Removal of the drums in the late 1960's and associated cleanup activities resulted in dispersion of contaminated soil to the east and to the south of the 903 Pad. The storage pad was covered with asphalt in 1969, and is no longer a source of resuspendable actinides. However, the initial spread of Pu- and Am-contaminated soil prior to the installation of the asphalt pad resulted in a plume of actinides in the surface soils extending to the east and southeast from the 903 Pad itself.

Other spills and releases have resulted in smaller areas where the surface soils are contaminated with different actinides (such as uranium [U] isotopes) over portions of the Site. In addition, naturally occurring U deposits may result in areas of elevated surface soil U concentrations as well. Actinide concentrations in surface deposits at the Site have been sampled and mapped, and the resulting data form the basis for actinide emission estimates developed as part of the work reported here.

Between 1989 and 1995, resuspension of actinide-containing soils and transport through the air pathway occurred primarily due to natural processes, such as rainsplash or wind erosion. Remediation of contaminated soils and waste-disposal areas at the Site began in

1995. Such activities disturb contaminated soils and result in additional airborne particulates. Future resuspension of actinide-containing material will occur due to both natural and anthropogenic activities.

1.2 Summary of FY99 Air Pathway Activities

FY99 air pathway studies have focused on three aspects of airborne actinide migration: emission of actinides into the air from contaminated soils or debris (resuspension), transport of airborne actinides from source areas at the Site to other parts of the Site or off Site (dispersion), and removal of actinide-contaminated particles from the air to soil or water surfaces (deposition). For resuspension, a detailed review of applicable emission data and estimation methods was performed, then the best methods available were chosen to estimate actinide resuspension from Site activities and surfaces. Emissions have been estimated for ongoing, natural resuspension mechanisms at the Site, and emission estimation methods have been identified for anthropogenic activities, such as remediation or decontamination and decommissioning (D&D) activities, or prescribed burns. The emission estimation methods will be used in the future to estimate emissions from various potential future scenarios at the Site, including scenarios involving D&D and remediation activities.

In parallel with the emission estimation activities, a model was developed that can be used to simulate dispersion and deposition of actinides for a variety of emission events or scenarios. An annual scenario was modeled representing the "chronic" resuspension of actinides. Airborne actinide concentrations due to these ongoing emissions were estimated at a variety of locations on and around the Site. The airborne concentrations were also converted to dose units using U.S. Environmental Protection Agency (EPA) conversion factors. Annual deposition of actinides that have become airborne due to chronic resuspension mechanisms also was estimated at locations in and around the Site.

The uncertainty associated with the model and emission estimation methods was explored through a series of "sensitivity analyses" that varied factors that affect emissions, dispersion, or deposition. These analyses were used to indicate which of the input factors have large effects on model estimates and which have more modest effects. These results will be used to refine model input data in the future. In addition, a limited comparison was performed using actinide concentration data collected at Site ambient air monitors.

This report discusses the air pathway work performed and the results of the analyses. The report is organized into four additional sections. Section 2.0 discusses the emission estimation task. Section 3.0 reports on the dispersion and deposition model formulation and results of the modeling analysis for the chronic resuspension scenario. Section 4.0 discusses the sensitivity analyses and comparison runs. Finally, Section 5.0 summarizes the FY99 work and presents recommendations for the next phase of the air pathway effort. A bibliography is provided in Section 6.0 that documents the scope of the literature review performed to identify resuspension mechanisms and factors.

2.0 EMISSION ESTIMATION

The focus of this task was to review the research that has been conducted on naturally occurring particle resuspension and to develop methods for estimating the particulate actinide resuspension rate from contaminated soil areas at the Site. The emission estimates from particle resuspension provide the basis for predicting airborne activity concentration (i.e., picocuries per cubic meter of air [pCi/m^3]), airborne dose (in millirem [mrem]), and activity deposition on ground or water surfaces (in picocuries per square meter per year [$pCi/m^2/yr$]) of the various actinides, as described in Section 3.0 of this report. An additional objective was to identify emission estimation approaches for remediation-related activities, prescribed burning, and building demolition, for future air dispersion modeling and hypothetical risk evaluation.

Risk-related studies typically are focused on either chronic or acute impacts, depending on the contaminant of interest. In this study, the objective was to develop emission estimates that support the assessment of chronic resuspension effects. These emission estimates were then input to a dispersion model to produce long-term predictions (annual). The dispersion model selected as most appropriate for the study operates on an hourly time step, so the temporal resolution of the factors influencing the emission rate was constrained to an hourly period.

2.1 Review of Resuspension Research

An extensive review of the recent scientific literature was conducted to provide a basis for developing emission estimates from natural resuspension of contaminated soil. This review encompassed both research conducted in the scientific community as a whole, and research specifically conducted at the Site. The databases searched electronically for relevant articles were: National Technical Information Service (NTIS), EI Compendex, Energy SciTec, Georeferences, Dissertation Abstracts Online, Enviroline, Pollution Abstracts, and Inside Conferences. A bibliography of the articles and reports reviewed for this task is provided in Section 6.0 of this report.

2.1.1 Summary of Past Research

Over the past several decades there has been a significant amount of research on particle resuspension by wind (EPA, 1983). This research may be categorized into three areas, each reflecting different theories and techniques for predicting particle resuspension:

1) Wind erosion research by agricultural scientists;
2) Particle resuspension research related to evaluating the potential risks due to inadvertent soil contamination by radioactive materials; and
3) Emission rate or emission factor research conducted for the purpose of establishing techniques for estimating fugitive dust emissions.

Measurements of resuspension have typically been expressed in terms of a resuspension factor or resuspension rate. The resuspension factor is defined as the ratio of contaminant concentration in air (just above the contaminated surface) to the contaminant concentration on the surface (the factor is expressed in units of inverse meters [m^{-1}]). The resuspension rate represents the fraction of the surface contaminant removed in a unit time (expressed in units of inverse seconds [sec^{-1}]). The range in reported resuspension rates and factors varies by 8 to 11 orders of magnitude, respectively, reflecting the variety of experimental techniques and environmental conditions involved (Sehmel, 1984).

From field studies, resuspension has also been defined in terms of a resuspension flux (mass resuspended per unit time per unit area). The flux units are ideal for use in dispersion modeling of contaminated areas.

The following paragraphs provide an overview of each of the main areas of resuspension research.

Agricultural Wind Erosion—Research in particle resuspension dates back to the 1940's, when the focus was on predicting soil loss from agricultural fields and developing methods of reducing soil erosion. These early studies also focused on the movement of sand in desert areas. As a result of the agricultural research, a wind erosion equation was developed that estimates annual soil loss from cropland as a function of various field, soil, and climatic factors. The equation predicts the total amount of soil eroded, not differentiating between various particle sizes (i.e., small particles that become airborne and are transported with the wind, and larger particles that roll along the surface). With an emphasis on the movement of total soil mass, little was then understood about the movement of fine particles.

The process of saltation is most relevant to these early studies, as saltation usually accounts for the movement of the greatest mass (Nicholson, 1988). With saltation, millimeter-sized particles are propelled along the surface by the wind. As these large particles impact the soil they knock loose smaller particles, which then become entrained into the main air stream by turbulence.

In the 1970's research in the suspension of agricultural soils resulted in an estimation technique based on ambient wind speed and soil size distribution (EPA, 1983). This technique, originally developed by Dr. Dale Gillette, incorporates the effect of saltation and is further discussed in Section 2.3. Agricultural research during the 1970's also resulted in the U.S. Department of Agriculture developing a wind erodibility classification scheme for the predominant soil textural classes.

Resuspension of Radioactive Material—More recent research in particle resuspension has focused on the resuspension of radioactive materials inadvertently deposited on environmental surfaces. In contrast to the agricultural research, the motivation in conducting this research has been on developing a better understanding of

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the mechanisms involved, particularly with respect to inhalable particles, such that the health hazards associated with the contamination may be assessed. Many of these studies involved experiments in which tracer materials were deposited over soil areas and measurements were taken to determine resuspension by wind. Other studies have involved the direct measurement of the contamination in both the soil and near surface air. In all these studies the contaminant sources are represented by surface concentrations (e.g., mass per unit area or curies per unit area).

An extensive amount of research has focused on the spread of contamination from the accident that occurred in 1986 at the Chernobyl nuclear power plant. Various radionuclides were released, primarily in particulate form. In a study of dry deposition in village areas around Chernobyl, it was observed that agricultural activity in the form of sowing and harvesting of crops did not lead to a significant spread of contamination into the nearby village (Kashparov, et al., 1994).

The effect of Chernobyl has been observed at locations throughout Europe. Long-term radionuclide resuspension factors were calculated for a semi-rural environment in Germany (Rosner, et al., 1997). The resuspension factor (concentration in air divided by the deposition per square meter of ground area) was observed to increase by about a factor of three for cesium-137 (Cs-137) and by about 40% for Pu, due to the deposition of contamination from Chernobyl (the influx of fresh material from Chernobyl being quite evident during the year of the incident). After the effect of Chernobyl had passed (approximately three years later), an essentially constant Cs-137 and Pu resuspension factor of 10^{-9} m^{-1} has been observed. This observation is noted as confirming Pu and Cs-137 empirical models of the time-dependency of contaminant resuspension, which predict an approximately constant resuspension factor, once weathering of the deposit has occurred. (In comparison, a range in resuspension factor of 10^{-13} to 10^{-10} m^{-1} has been calculated in the past for areas near the 903 Field sampler on Site.)

Other studies of surface contamination and atmospheric concentrations in the 30 km exclusion zone around the Chernobyl plant indicate the importance of advection from upwind sources on measured air concentrations, as well as the influence of rainfall (Garger, 1994). For a limited sampling period four months after the incident, air concentrations were found to be dependent on advection of contaminated material from upwind. Lower concentrations were observed after a period of rainfall and the presence of a moist soil surface.

Air Pollutant Emission Estimation—A third area of research initiated in the mid 1970's and continuing through the 1980's has been the investigation of fugitive dust emissions from sources such as outdoor storage piles, paved and unpaved roads, and soil areas exposed due to grading and excavation. The primary objective of this work has been to develop emission factors, which are estimates of the rate at which a pollutant is released to the atmosphere as a result of some activity, divided by the unit measure of that activity. An example of an emission factor is the grams of dust emitted per mile of

vehicle travel on unpaved roads. As presented in Section 2.5 of this report, the outcome of some of this research will be employed in estimating emissions due to future remediation, building demolition, and prescribed burning events on Site.

2.1.2 Summary of Research Conducted at RFETS

A considerable amount of resuspension research has been conducted at the Site over the years. This work was principally performed by and under the direction of Gerhard Langer. The general topics investigated were the:

- Effect of wind speed on resuspension (including the significance of saltation as a resuspension process on Site);
- Vertical distribution of Pu and dust concentration;
- Change of resuspension rate with time;
- Effect of rainsplash;
- Effect of various surfaces on resuspension (vegetated, litter only, dry soil, moist soil);
- Airborne Pu activity distribution;
- Airborne soil particle size distribution; and
- Activity in vegetation vs. activity in the underlying soil.

Past research on Site has also led to the development of a Site-specific resuspension rate, which was estimated to be 2×10^{-12} sec^{-1} for the entire 903 Field (area adjacent to and directly east of the 903 Pad) (Langer, 1991). This parameter has been used in developing actinide emission estimates for the contaminated soil areas on Site in support of the Site's annual reporting on radiation dose to the public required under Title 40 of the Code of Federal Regulations (CFR), Part 61, Subpart H and Colorado Air Quality Control Commission Regulation No. 8, Part A, Subpart H.

Additional work was performed in 1993 by Dr. Chatten Cowherd of Midwest Research Institute (MRI). This work was conducted within Operable Unit 3 (OU3), located just to the east of the Site, using a portable wind tunnel. While OU3 is located outside of the Site boundary, the research is relevant given the similarity in surface characteristics to the contaminated soil areas on Site.

2.2 Conceptual Model of Particle Resuspension at RFETS

This section describes natural resuspension processes that appear to contribute to wind erosion at the Site. Understanding these processes provides a conceptual framework for evaluating the technical bases for the emission estimation approach selected.

2.2.1 Emission Source Characteristics

Actinides occur in Site surface soils due to past spills and releases, as well as the natural distribution of uranium in some area soils. As described above, the most extensive

release was from a contaminated area known as the 903 Pad, where drums of waste cutting oil, containing Pu particles, were stored. Over time the drums leaked oil onto the underlying soil. The drums were removed in the late 1960's and the area was covered with gravel and asphalt to immobilize the Pu-contaminated soil particles.

During site preparation for applying the asphalt, occasional high winds swept across the uncovered area. This suspended contaminated dust generally settled a short distance to the east of the 903 Pad area. This adjacent area, known as the Pad Field, was subsequently covered with topsoil obtained off Site, and revegetated to control resuspension of the Pu particles. During this process, the fugitive Pu particles were mixed into approximately the top 20 centimeters (cm) of new topsoil, making some Pu available for resuspension.

The contaminated soil area resulting from the 903 Pad remediation constitutes the largest and most significant "source" area for the actinide migration study. Other smaller releases of U, Pu, and Am isotopes have resulted in isolated areas above background concentrations at other locations on Site.

Soil isopleth maps showing the estimated distribution of actinide activity have been developed from soil sampling conducted on Site. These maps provide the spatial "source strength" that forms the basis for the emission estimates developed in this task, as well as for annual regulatory analyses of Site emissions, as required by 40 CFR 61, Subpart H. The isopleths characterize soil activity levels for each actinide in units of pCi per gram (pCi/g) of soil. While the isopleth maps associate geographic areas with certain activity levels, it is understood that the spatial distribution of contamination is not necessarily homogeneous within any given level. The true distribution is not known. However, for the purposes of conducting the dispersion modeling it has been assumed that the soil actinide concentration within the area defined by a given isopleth is sufficiently uniform to yield representative air concentrations.

The Pu particles in the cutting oil that leaked at the 903 Pad were small (< 3 micrometers [μm] diameter). Once in contact with the soil, however, the Pu particles became attached to soil particles. Experimental data from the Site (Langer, 1986) and elsewhere (Shinn, 1999) indicate that most of the airborne Pu activity is carried on larger particles (particles >15 μm in diameter). These larger particles are generally aggregates (i.e., the Pu particles are attached to soil aggregates, rather than to primary soil particles). Therefore, the transport of Pu is dependent on the soil (aggregate) particle properties, and not the properties of the individual Pu particles.

Over several years Langer collected dust and Pu activity concentration data from a sampling platform located in the "East Field" (field located directly east of the Pad Field). Data were collected at three heights above ground (1, 3, and 10 meters [m]). For each height, the concentration data were reported for three particle size fractions: respirable

(< 3 μm), inhalable (3-15 μm) and coarse (> 15 μm). In general, Langer found that the Pu activity is approximately proportional to the dust mass, with about 70% of the Pu activity concentration residing on the coarse particles, which represent 60% of the total dust concentration.

Past research at the Site indicates that coarse particles carry most of the uranium activity as well (Langer, 1987). Consequently, the activity distribution among various soil particle size classes was considered the same for each of the five isotopes studied (Pu-239, Am-241, U-233/234, U-235, and U-238).

The particle mass fraction and actinide activity fraction data are important inputs to the deposition modeling. The measured data that were used in the modeling are shown in Table 2-1. The particle density and effective geometric diameter affect gravitational settling, with more dense particles depositing closer to their emission area than less dense particles. How the particle density varies by size category is not known. Therefore, for the initial modeling, a value of 2.5 grams per cubic centimeter (g/cm^3) was selected, as representative of a mineral topsoil (Brady, 1974).

Table 2-1. Particle Size Distribution Data Used for Dispersion Modeling

Particle Size Category	Particle Diameter (μm)[a]	Particle Density (g/cm^3)	Particle Mass Fraction[b]	Plutonium Activity Fraction[b]
1	3 (<3)	2.5	0.17	0.04
2	10 (3-15)	2.5	0.23	0.19
3	15 (>15)	2.5	0.60	0.77

[a]Category 1 represents particles with diameter < 3 μm.
Category 2 represents particles with diameters between 3 – 15 μm, inclusive.
Category 3 represents particles with diameters > 15 μm.
[b]Values at the measurement height of 1 m (Langer, 1987).

Notes:
μm = micrometers
g/cm^3 = grams per cubic centimeter

The particle size distribution actually is a function of wind speed. The higher the wind speed, the greater the fraction of larger particles. However, insufficient data are available to characterize a change in distribution with wind speed, so the distribution has been assumed constant.

A fundamental question to address when developing an actinide emission estimation approach is whether resuspension would occur from both bare soil areas and areas of vegetation. Since the completion of the 903 Pad remedial action, the Pad Field has re-vegetated. Furthermore, the vegetation existing in the Pad Field today is more dense than it was during the period of wind tunnel studies conducted during the 1980's (Langer,

1999). In the present study, a visual survey of the contaminated soil areas on Site has been made, and has revealed very few bare soil areas. Fairly dense vegetation presently occurs throughout the source area (mostly grasses of varying heights). Where small, bare areas do exist, they appear to be protected from the full force of the wind by the surrounding vegetation (grass) canopy.

To address the question of whether saltation could be occurring from small areas of bare soil, Langer conducted studies using laser beam and acoustic particle detector technology. A laser beam was used to graze the soil surface and detect impacting large particles and bursts of numerous small particles. No particles were observed until winds exceeded 16 meters per second (m/s) (35 miles per hour [mph]), but Langer noted that even then no clear evidence of saltation was found. In a second experiment, an acoustic particle detector was placed underground within a bare soil surface area. The objective was to catch large particles resuspended by the wind that then fall back to the ground. Even under conditions of high wind, no clear case for saltation could be made with this study either. Consequently, given the results of these experiments and the current surface characteristics of the contaminated soil areas, the more significant source of chronic, natural contaminated soil resuspension on Site may be dust-laden vegetation and litter, rather than exposed soil surfaces.

The unpaved roads occurring through the contaminated areas were not considered bare soil areas for this study because the roadway dirt was assumed to be relatively uncontaminated. Roads in the immediate area of the 903 Pad were surfaced with soil and road base brought in from off Site during the 1980's.

Past research at the Site has shown that, on a mass basis, actinide activity in the soil attached to vegetation and litter surfaces is less than the activity of the underlying bulk soil. Langer (1986) found that soil washed from vegetation carried only 20% the activity in the underlying soil (on a pCi/g basis). Other measurements at the Site indirectly support dilution relative to the underlying soil activity (Little, et al., 1980; Arthur and Alldredge, 1982; Langer, 1986).

The apparent dilution is likely due to the advection and deposition of "clean" particles from upwind areas, and may also be partially a function of the size distribution of particles on plants. As previously mentioned, activity is not carried uniformly across different particle-size classes. Further, different size particles will be preferentially resuspended, deposited on vegetation surfaces, and retained on vegetation surfaces. For the initial modeling, it was assumed that the particle activity on the vegetation is the same as the activity of the underlying soil (this is a conservative initial position, given the limited data available).

2.2.2 Mechanisms for Release to Atmosphere

Resuspension at the Site is an ongoing phenomenon that is thought to be episodic in nature and influenced primarily by meteorological variables (wind speed and rainfall);

particle and soil properties (moisture level and particle density); and surface characteristics (density and type of vegetative growth, and snow cover). The general mechanisms whereby particles may be naturally resuspended are:

- Wind erosion of exposed soil surfaces;
- Wind erosion of contaminated soil from vegetation surfaces;
- Wind erosion of contaminated soil on litter;
- Wind erosion of decaying litter;
- Rainsplash; and
- Burning of contaminated vegetation by grass fires.

Exposed Soil—The role of saltation and direct wind erosion in chronic, natural resuspension is considered insignificant given that little bare soil area appears to exist in the contaminated areas on Site. As described in the previous subsection, the source of resuspended contaminated soil appears to be the dust-laden vegetation and litter. Grass is the predominant type of vegetation in the contaminated area, though trees and shrubs are present along the creek beds. (The primary grass in the reclaimed area is smooth brome, in the mesic grassland it is western wheat grass, and in the xeric grassland it is big blue stem and mountain muhley.) Plant litter is not readily accessible for resuspension because it is shielded from the wind by the overlying grass cover.

Vegetation and Litter—The means by which the contaminated soil becomes attached to the grass and decaying vegetation (litter) is a key step in the overall process of resuspension. In the case of Pu and Am, plant uptake is not a significant factor (Baes, et al., 1984), leaving deposition of soil particles from the wind, and rainsplash, as the dominant mechanisms. While U isotopes are more readily absorbed, the U in attached soil probably also represents a larger reservoir of resuspendable material than U in plant tissues.

Previous research on rainsplash has shown this to be a significant means of transferring soil to vegetation, as discussed later in this section. A complicating factor with rainsplash, however, is the vegetation density. If an area is densely vegetated, then rain will impact the vegetation and not bare soil. This is thought to be the case over much of the Site source area.

Scanning electron microscopy of the surface of grass blades indicates that most grass blades are covered with fine hairs that serve to enhance particle deposition and retention (Langer, 1987). Particles would become available as the grass hairs decay and fall off, and as the blades brush against each other in the wind.

The resuspension of particles from grass blades has been investigated in a set of simple tests (Langer, 1987). Filtered air was directed over a few blades of grass placed on a screen in a small laboratory wind tunnel. An optical particle counter monitored the particles resuspended from the blades. At the low, 10-m equivalent wind speed of 2.3

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m/s (5 mph), particles in the 0.2 to 12 μm range were resuspended, with 95% of the resuspended particles having diameters less than or equal to 1 μm (aerodynamic equivalent diameter). At a 10-m equivalent wind speed of 11 m/s (25 mph), the size distribution of the particles shifted, with only 40% of the resuspended particles having diameters less than or equal to 1 μm.

Tests were also conducted in which the grass blades were mechanically flexed. A membrane filter collected the dust for microscopic examination. For these tests, the release of particles greater than 10 μm was dominant, with a median diameter for the released particles of 20 μm and a maximum of 40 μm (Langer, 1987).

Rainsplash—Particle resuspension from rainsplash also has been investigated on Site in response to previously unexplained Pu sampling data from the Pad Field. In the spring of 1981, there were week-long periods when the field was continuously wet due to rain, but there was no significant reduction noted in airborne monitored Pu concentrations. Initially, a laboratory wind tunnel was constructed to simulate raindrops splashing on soil under controlled conditions. The results of this experiment showed that soil particles become incorporated into aerosols by rainsplash if a thin water layer exists on the soil surface. The airborne soil particles are the residual particles that remain upon evaporation of the small satellite drops that form upon impaction of the larger, initial raindrops (Langer, 1987).

The resuspension process due to rainsplash was confirmed in the field. A total of 2,500 soil particles were observed to be resuspended from 1,000 5-millimeter (mm) raindrops impacting a bare soil area (Langer, 1987).

Assessing the influence of rainsplash is complicated by the fact that rainfall also serves as a mechanism for removing particles from the atmosphere. Its importance as a mechanism for resuspension is dependent on the intensity and amount of rainfall, and on the type of surface the rain drops impact (bare soil vs. vegetation). As mentioned previously, with the increase in vegetation density in the contaminated soil area since the 1980's, rainsplash is probably not a significant mechanism for the transfer of soil to vegetation. The direct resuspension of particles from vegetation, due to rainsplash, has not been evaluated.

Grass Fires—Contaminated soil also can become airborne as part of the ash from grass fires (either prescribed burns or unplanned fires). Particulate matter released from fires has been the subject of study by both the U.S. Forest Service (USFS) and the U.S. Bureau of Land Management (BLM). The USFS has developed an emission factor-based approach to estimating emissions from fires (grams of particulate released per gram of vegetation burned). The BLM has developed a model for estimating particulate concentrations based on estimates of the heat released from the fire and the associated plume rise. The key factors in estimating actinide emissions from fire include total fuel

mass available per unit area, fraction of total fuel mass that actually burns, mass of contaminated soil on the vegetation/litter, and the activity of that soil.

2.3 Soil Resuspension Emission Estimation Approaches

Based on the review of literature on particle resuspension and the review of experiments conducted on Site, certain methods for estimating emissions have been identified as potentially useful. The attributes and merits of these approaches are described in this section. The approach selected for estimating emissions from the natural resuspension of contaminated soil on Site is then presented.

2.3.1 Wind Erosion Models

Wind erosion models for estimating particulate emissions from open areas have been documented in EPA guidance for over a decade (EPA, 1985; EPA, 1988). Two types of models have been identified based on the erodibility of the surface material. Nonhomogeneous surfaces containing nonerodible elements (such as stones and vegetation) are characterized as having a finite availability, or "limited reservoir," of erodible material. These surfaces have high threshold wind speeds for wind erosion, and particulate emission rates tend to decay rapidly during an erosion event. Bare soil surfaces, such as agricultural soil, are characterized as having an "unlimited reservoir" of erodible material. These surfaces have low threshold wind speeds for wind erosion, and particulate emission rates are relatively independent of time at a given wind speed. Both models are based on the concept of a threshold wind speed, which is the wind speed needed to initiate erosion.

The basic expression for the unlimited reservoir model is:

$$E = \chi\left(\frac{u}{u_t}\right)^{\gamma}$$

where:

E is the particulate emission flux ($g/m^2/s$);
χ is a soil suspension calibration constant;
γ is a wind speed exponent;
u is the ambient wind speed (at 10 m); and
u_t is the threshold wind speed (at 10 m).

The threshold wind speed (u_t) is generally provided by the log wind speed profile:

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$$u_t(z) = \left(\frac{u_*}{0.4}\right) \ln\left(\frac{z}{z_0}\right)$$

where:

z is the height above the ground (10 m);
z_0 is the surface roughness length; and
u_* is the threshold friction velocity.

An empirical expression for u_* is given by the equation:

$$u_* = 65.5315\, p^{0.417672}.$$

In this equation "*p*" is the mode of the surface, dry aggregate size distribution. The guidance suggests a simple hand-sieving test of the uncrusted surface soil to determine the size distribution mode. Crusted surfaces are regarded as having a limited reservoir of erodible particles.

The basic expression for the limited reservoir model is:

$$E = 0.83 \frac{f\, P(u^+)(1-V)}{\left(\frac{PE}{50}\right)^2}$$

where:

E is the particulate emission flux ($g/m^2/s$)
f is the frequency of disturbance per month;
$P(u^+)$ is the erosion potential [$6.7\,(u^+ - u_t)$ for u^+ greater than or equal to u_t ; otherwise 0];
u^+ is the observed (or probable) fastest mile of wind for the period between disturbances;
u_t is the erosion threshold wind speed;
PE is Thornthwaite's Precipitation Evaporation Index; and
V is the fraction of contaminated surface vegetative cover.

A disturbance is defined as an action resulting in the exposure of fresh surface material. The fastest mile is the fastest sustained wind speed observed to "move" a mile of wind past a given point (e.g., a 27 m/s [60 mph] wind would need to be sustained for 1 minute to be considered as a fastest mile wind). Generally, values of fastest mile are less than the peak, instantaneous gust values observed at a given location.

These wind erosion models were not considered appropriate for the natural resuspension scenario under investigation in this study. The unlimited reservoir model is based on field measurements of highly erodible soils, such as pure sand. Such soils are not found

on Site, so the unlimited reservoir model is not applicable. Furthermore, based on previous soil aggregate particle-size mode determinations made at OU5 (EG&G, 1995) and the magnitude of the hourly average wind speed, no resuspension would be estimated from this approach, as calculations of the threshold wind speed were found to exceed 55 m/s (120 mph).

The limited reservoir model assumes that there is some periodic disturbance of the surface that exposes fresh material for resuspension. This implies some mechanical disturbance of the surface, which is not assumed to be the case for this study (but may well be a factor in future remediation or D & D scenarios). Natural processes such as seasonal freeze-thaw action, growth of new vegetation, and animal activity could be considered a disturbance, but quantifying such processes for input to the equation is uncertain.

2.3.2 Meteorological Flux Gradient Model

A model for estimating the vertical dust flux has been developed based on micrometeorological parameters (Anspaugh, L. P., et al., 1975; Shinn, et al., 1997). During periods of wind speed sufficient to produce a vertical flux of aerosol, the aerosol concentration at a height of 1 m appears universally to exhibit a power law distribution. While vertical flux is a function of height, a 1 m height is chosen because the aerosol concentration varies by only about +/- 20% between 0.5 and 2 m. The expression is:

$$F = \left(\frac{1}{\Phi}\right) u_* \, k \, p \, C$$

where:

F is the (dust or activity) flux ($g/m^2/s$);
ϕ is a stability correction term;
u_* is the friction velocity;
k is von Karman's constant (0.4);
p is the exponent of a presumed power-law distribution of concentration with height; and
C is the (dust or activity) concentration in the surface layer.

Representative data from the Site are available to calculate a value of p, which can be a fixed value for the purpose of estimating the dust flux (this assumes that the vertical dust profile has a consistent shape over the time period in question). To apply this method, however, concentration and friction velocity need to be on the same averaging basis. While Site data are available to estimate friction velocity on an hourly basis, such is not the case with dust concentration. Furthermore, adequate data do not exist for establishing a relationship between friction velocity and concentration at the Site, which otherwise could be used as an alternative.

2.3.3 Site-Specific Models

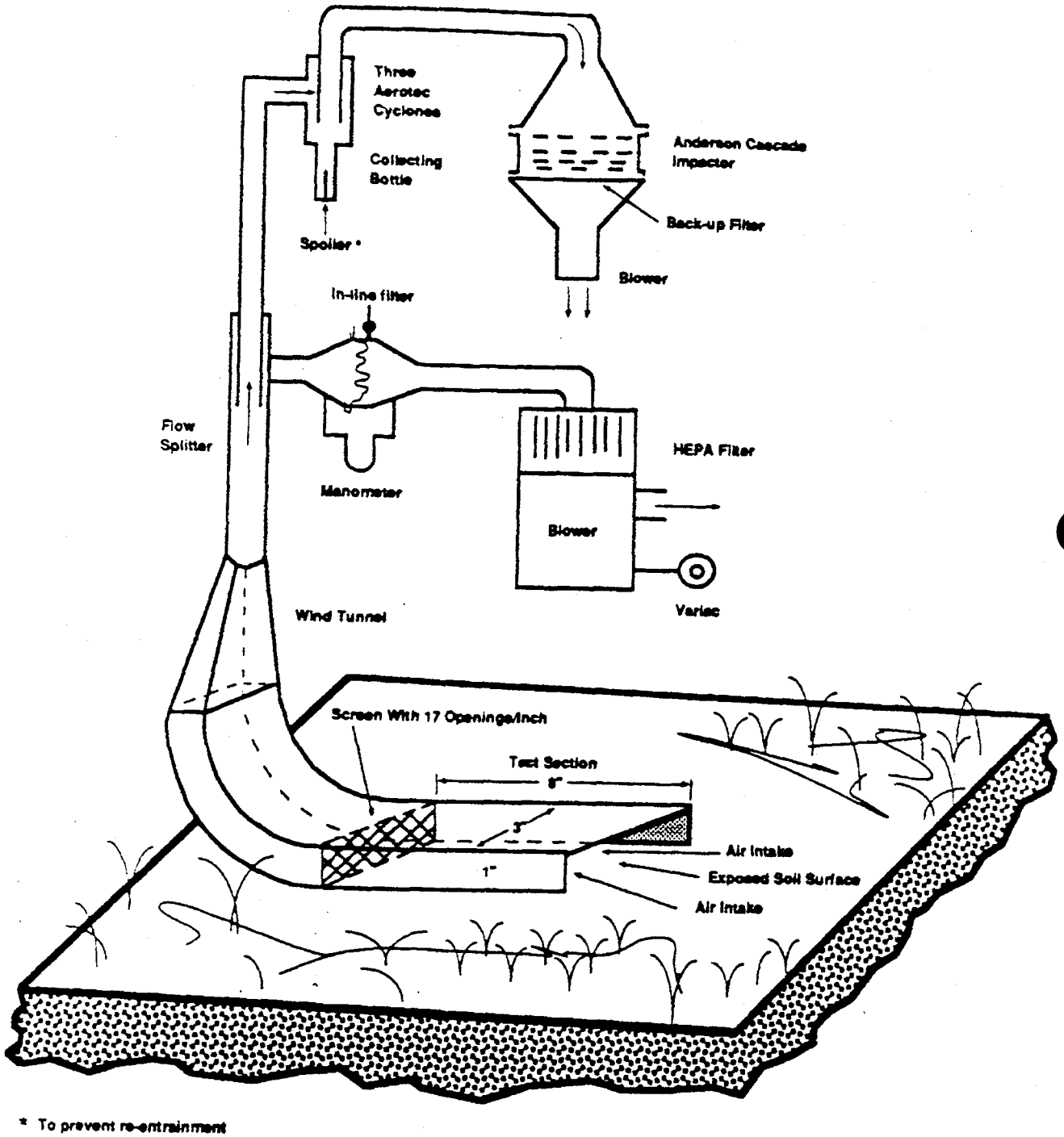
Over the years, several wind tunnel studies have been conducted on Site. The basic design of the wind tunnel used by Langer is shown in Figure 2-1. With this design, wind from the test section was drawn first through cyclones to remove the coarse (> 10 μm) particles and then through a cascade impactor to collect inhalable particles (< 10 μm). A backup filter trapped the respirable (< 3 μm) particles. The wind tunnel was not specifically designed for use over vegetated surfaces, as its test section was in some cases less than the height of the grass.

The most recent wind tunnel work at the Site was conducted at OU3 (just east of Indiana Street) in June 1993 (EG&G, 1994). This field experiment employed a wind tunnel design that has been documented in *Air/Superfund National Technical Guidance Study Series, Volume II, Estimates of Baseline Air Emissions at Superfund Sites* (EPA, 1989), and is shown in Figure 2-2. (Under contract to EG&G, this wind tunnel experiment was conducted under the direction of Dr. Chatten Cowherd of Midwest Research Institute). The test section height of this wind tunnel was approximately 15 times greater than the height of the wind tunnel used previously, and consequently it was better suited for obtaining measurements over vegetation.

For the OU3 study, the objective was to characterize resuspension under various levels of surface disturbance for both terrestrial and shoreline sites (locations along the shores of Standley Lake and Great Western Reservoir). Only a few tests were conducted for undisturbed terrestrial sites, which are the focus of the present study. While particles were collected from sampling at the undisturbed terrestrial sites, visible resuspension of soil particles was not observed for these cases, even at the flow capacity of the tunnel. This supports the conclusion that there is not an infinite reservoir of erodible particles at these locations.

A listing of the resuspension flux determined through the various wind tunnel experiments is provided in Table 2-2. The values are shown graphically in Figure 2-3. The plot reveals three distinct populations of data. It is unclear why the "East Field" fluxes determined in July 1982 are so much higher than the other data. In contrast, the fluxes established from the OU3 wind tunnel work are among the "lowest" population of data.

Figure 2-1. Langer Portable Wind Tunnel



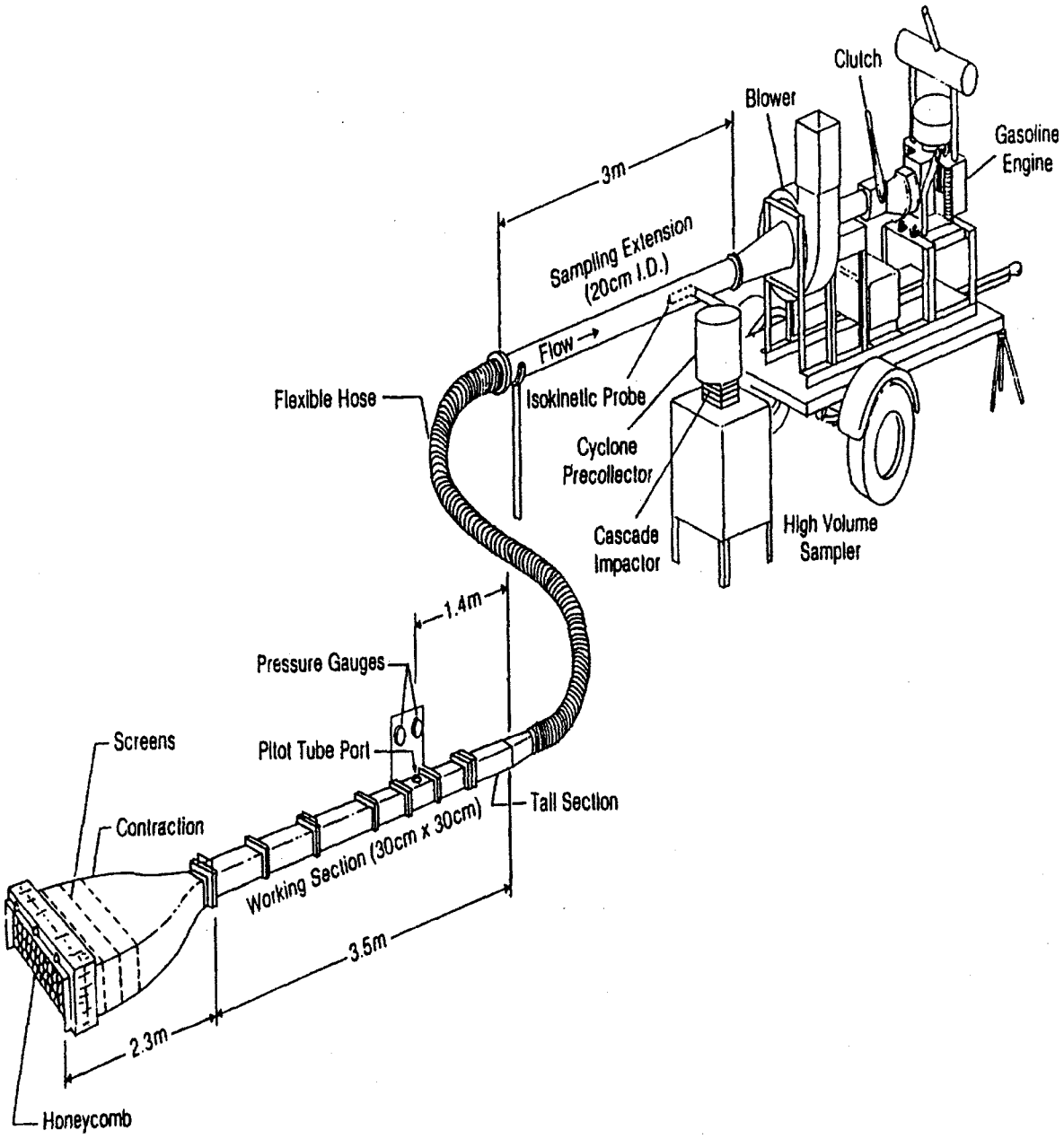


Figure 2-2. MRI Portable Wind Tunnel

Table 2-2. Compilation of Site Wind Tunnel Measurements Over Grass

Wind Speed[a] (m/s | mph)	Dust Resuspension Flux[b] ($g/m^2/s$)	Comments
9.9 / 22	1.65×10^{-5}	Area of sparse grass.
22.4 / 50	3.05×10^{-4}	--
22.4 / 50	3.95×10^{-3}	East Field data.
26 / 58	2.55×10^{-4c}	--
29.1 / 65	6.11×10^{-4}	--
29.1 / 65	5.73×10^{-3}	East Field data.
31.4 / 70	5.09×10^{-4c}	--
31.8 / 71	4.33×10^{-5}	From OU3 wind tunnel study.
31.8 / 71	6.50×10^{-5}	From OU3 wind tunnel study.
35 / 78	7.64×10^{-4c}	--
35.9 / 80	9.55×10^{-5}	Litter only. Grass cut to 0.5 cm.
35.9 / 80	2.29×10^{-4}	Litter only. Grass cut to 0.5 cm.
35.9 / 80	4.07×10^{-6}	Grass only. Soil wetted.
38.1 / 85	1.02×10^{-3c}	--
40.3 / 90	1.27×10^{-3c}	--
40.3 / 90	1.65×10^{-3}	--
40.3 / 90	6.24×10^{-3}	--
43.0 / 96	1.57×10^{-4}	From OU3 wind tunnel study.
44.8 / 100	1.30×10^{-4}	From OU3 wind tunnel study.

[a]10-m equivalent.
[b]Rates adjusted to 60-minute exposure.
[c]Estimated from plotted trend line (Langer, 1984).

Notes:
m/s = meters per second
mph = miles per hour
$g/m^2/s$ = grams per square meter per second

The reason for the overall higher resuspension fluxes exhibited from the earlier wind tunnel work performed by Langer was explored. Surface conditions at the undisturbed OU3 sites that were tested in 1993 appear to be similar to the surface conditions during data collection in the 1980's (i.e., the height and density of the grass surfaces studied appear comparable). The key reason for the differences in resuspension fluxes appears to be that the low test section height of the Site wind tunnel did not allow for a realistic, surface wind profile to be established. The test section wind speeds are therefore considered to be artificially high (Langer, 1999).

The flux values in Table 2-2 have been adjusted to reflect the fact that the flux of resuspension is not constant over time, but instead decreases exponentially as a function of surface erodibility. Most of the wind tunnel data collected reflect the resuspension flux after a 1-minute exposure. It would be overly conservative to apply this flux to an hourly time period (the time step used by the dispersion model). Therefore, a means of adjusting the 1-minute data to a 60-minute exposure was needed. Langer conducted a test

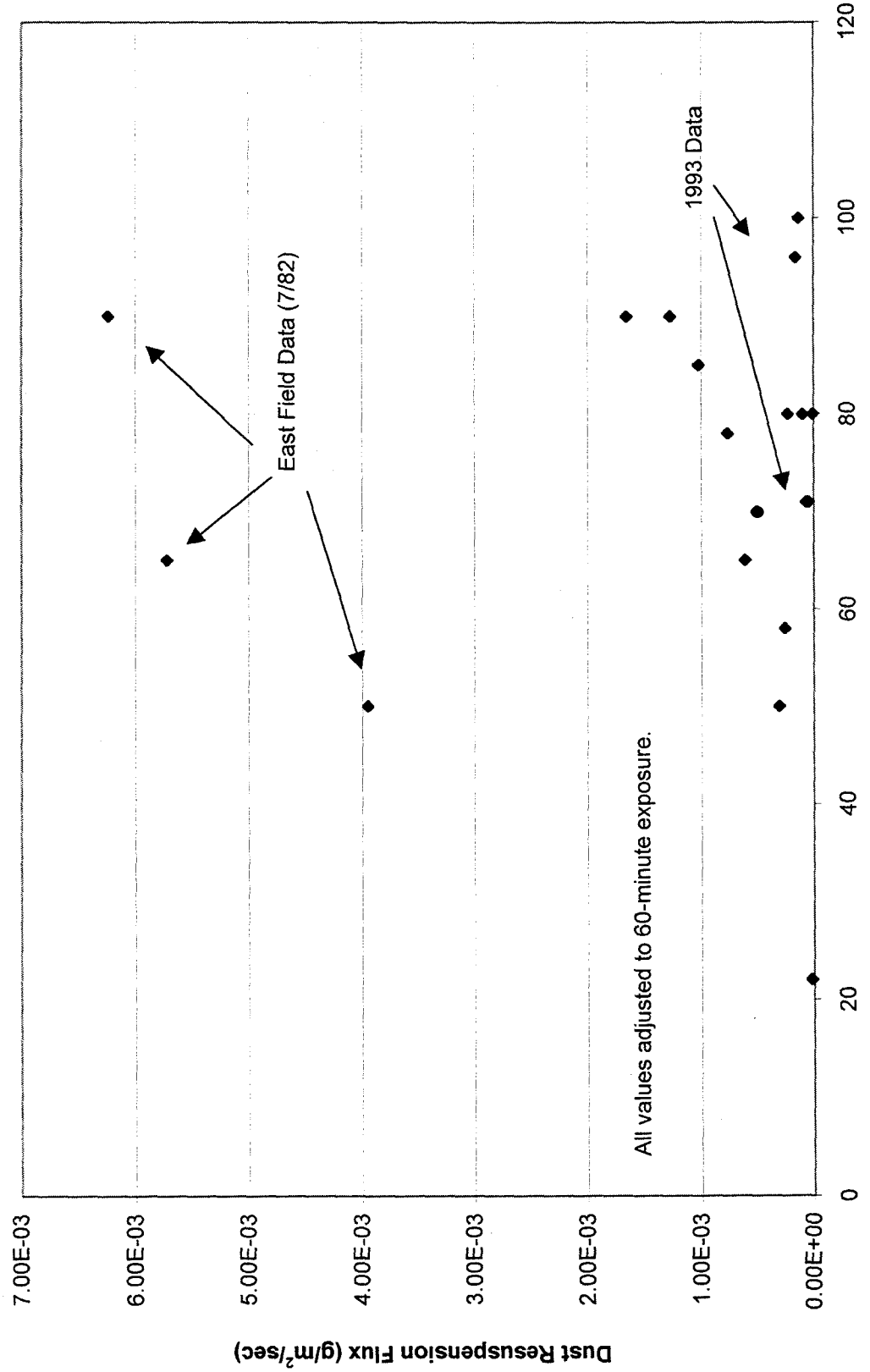


Figure 2-3. Site Dust Resuspension Flux from Grass

over a sparsely vegetated area to explore this issue (Langer, 1984). The dust resuspension flux was determined after 1, 15, and 60 minutes for the same location. The rate after 60 minutes was observed to be only about 1% of the flux after 1 minute. The fluxes and exposure times were plotted and a power fit to the data was determined, providing a means of adjusting the empirically-determined 1-minute resuspension fluxes to fluxes reflecting a 60-minute exposure.

2.3.4 Selected Approach for Estimating Natural Resuspension

Although the number of data points is extremely limited, the 1993 OU3 wind tunnel data set is considered the most representative of current conditions on Site. A power fit to the data produced the following expression:

$$E = 2 \times 10^{-9} (U^{3.014})$$

where:

E is the total particulate emission rate ($g/m^2/s$); and
U is the 10-m wind speed (m/s).

It is necessary to put this result in perspective. In general, both field studies using inert tracers and wind tunnel studies conclude that resuspension increases as some power of wind speed (Sehmel, 1984). Values of the power have been reported to range from 1 to 6 for particles in the respirable size range. Agricultural research (focused on particles in the nonrespirable size range) has shown that fields erode as a function of wind speed raised to the third power (Sehmel, 1984). Measurements made at the GMX site of the Nevada Test Site, which is described as having a "typical desert pavement protected by native vegetation, and only slightly erodible," have indicated a dust flux proportional to the wind (friction) velocity raised to the third power ($u_*^{3.09}$) (Anspaugh, et al., 1975). Therefore, the power fit to the Site data is similar to what has been identified in other studies.

A concern with the use of the early wind tunnel data collected on Site is that the Langer wind tunnel was never formally calibrated. The MRI wind tunnel used in the OU3 field experiment had been calibrated, however. A power fit to Langer's data (data points in the "middle" region of the plot shown in Figure 2-3) also produced a third power relationship, but with a different constant. Recall that the test section height of the MRI wind tunnel was approximately 15 times greater than the height of the Langer wind tunnel, and was consequently better suited for obtaining measurements over vegetation (hence, the OU3 data are considered much better defined in terms of the wind profile) (Langer, 1999). The constant in the expression derived from the OU3 data is thought to reflect some of the calibration lacking in the Langer data.

In the selected approach, estimated emission rates were set equal to zero if snow cover was present. To establish the presence of snow cover, an hourly record of precipitation type was obtained from the National Weather Service station at Denver International Airport. Solar radiation measurements in terms of global horizontal radiation and ground-reflected global radiation were obtained for the year of interest from the National Renewable Energy Laboratory (NREL) site in Golden, CO. Measurements from this site are considered to be representative of conditions at Rocky Flats. Surface albedo was calculated as the ratio of ground-reflected radiation to global horizontal radiation. A clear correspondence between the precipitation-type data and the calculated values of albedo was observed. Emission rates were set equal to zero for those hours when the calculated albedo was greater than 0.40 (Oke, 1978).

Figure 2-4 shows the temporal variation in estimated emissions (using 1996 meteorological data) based on this approach. (Note that 1996 meterological data have been used in this study because the 1996 data show fewer missing values than 1997 or 1998, for parameters needed for the emission estimation and modeling). The median dust resuspension flux for the year was calculated to be 8.7×10^{-8} g/m^2/s, and the range was calculated to be 2.0×10^{-9} to 3.2×10^{-5} g/m^2/s. Ambient Pu concentrations measured at a sampling location located just east of the 903 Pad during 1996 appear to support the emission estimates (similar temporal trends are observed). (A comparison of ambient Pu data with model results and assumptions is discussed in Section 4.1 of this report).

2.4 Emission Estimation Method for Wildfires

Another "natural" resuspension mechanism is the release of actinides contained in vegetation or attached to vegetation surfaces when the vegetation is burned. Fires may be planned or unplanned. Unplanned fires may occur at the Site due to lightening strikes or due to accidental ignition of flammable vegetation by other means. This section discusses emissions from unplanned fires. Emissions from planned fires (prescribed burns) are described in Section 2.5.3.

Federal land management agencies, including the USFS and the BLM, have studied fire emissions extensively and have developed a variety of tools to estimate particulate emissions and dispersion from fires. The EPA has also published particulate matter emission factors for fires in *Compilation of Air Pollutant Emission Factors, AP-42* (EPA, 1995a).

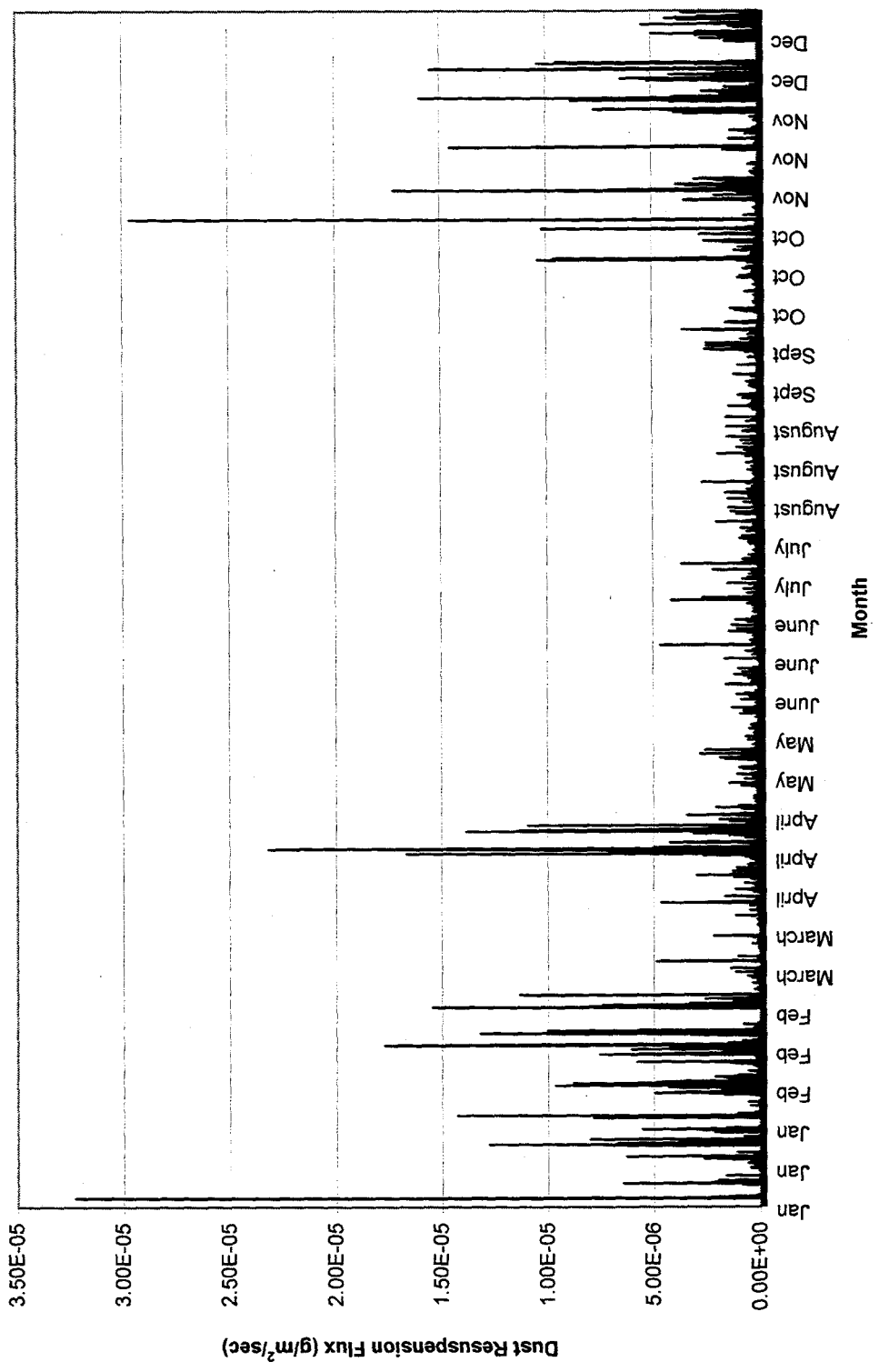


Figure 2-4. Temporal Variation in Dust Resuspension Flux

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2.4.1 Particulate Emission Estimates

Particulate emissions from fires vary with the fuel type (e.g., grass, shrubs, trees, etc.) and with the fuel loading (the mass of fuel per unit area). The nature and amounts of pollutants are thought to be directly related to the intensity and direction (relative to the wind) of a wildfire, and indirectly related to the rate at which the fire spreads. The latter may be influenced by a variety of variables, including the weather, fuel parameters, and topography (EPA, 1995a).

Particulate emissions from wildfires can be calculated from the amount of fuel that is burned per unit area, the rate at which fuel is consumed, and an emission factor relating the mass of particulates released to the mass of fuel burned:

$$Q = \left(\frac{(EF)(FL)(CF)}{T} \right)$$

where:

Q is the particulate matter emission rate (mass particulates emitted per unit area per unit time; i.e., grams per square meter per second [$g/m^2/s$]);
EF is the emission factor (in mass particulates emitted per mass fuel burned);
FL is the fuel loading (mass fuel per unit area);
CF is the fuel consumption factor (proportion of available fuel consumed in the fire); and
T is the total duration of fire (seconds).

This formula estimates particulate emissions from wildfires in the units required by the dispersion model used in this study. The emissions from the entire fire would be a function of the total area burned.

2.4.2 Emission Factor

Particulate emissions from fires have been estimated by a number of researchers. Emission factors vary depending on the fuel type, the rate of energy release (fire intensity), and the various fire phases (flaming, glowing, smoldering, etc.). Ward, et al. (1996) give a range of emission factors for African grassland savanna ecosystems. EPA gives emission factors for a variety of fuel types and fire phases in AP-42, Tables 13-3 and 13-4, based on prescribed burning data (EPA, 1995a). AP-42 also gives emission estimates for wildfires, but the estimates are based on typical vegetation cover in broad regions of the United States and are not applicable to Site conditions.

Vegetation over most of the Site, particularly areas with actinide contamination, is composed primarily of grassland and, to a lesser extent, shrubland communities.

Emission factors for grass or shrub fires range from approximately 2 g of particulate matter per kilogram (kg) of vegetation burned to around 23 g/kg based on the previously cited references. Because various researchers have measured particulates in different ways, the factors are not strictly comparable (data may represent particulate matter less than 2.5 μm in diameter [$PM_{2.5}$], less than 10 μm diameter [PM_{10}], or up to 30 μm diameter). Generally, the smaller g/kg values represent smaller particles and the larger values represent total particulates.

Smoke from fires is a complex mixture of carbon, tars, liquids, and different gases. This open combustion source produces particles of widely ranging size, depending to some extent on the rate of energy release of the fire. The turbulent nature of high intensity fires, for example, may entrain larger particles, composed of ash and partially burned plant matter that are not produced by lower intensity fires. As a result, high intensity fires often show a bimodal particle size distribution, with peaks near 0.3 μm and exceeding 10 μm. (EPA, 1995a)

For future wildfire scenarios, particulate emission factors consistent with the above information will be used. The emission factor chosen will depend on whether total airborne concentrations, respirable size fractions, or deposition values are desired, as well as what type of vegetation will be burned and what weather conditions are to be simulated.

2.4.3 Fuel Loading and Consumption Factor

Fuel loading may be expressed in several ways—as the mass of combustible material that will be consumed in a wildfire under specific weather conditions (available fuel), as the mass of all combustible material that would burn under the most severe weather and burning conditions (total fuel), or as the amount of larger, woody material that would remain even after an intense fire (potential fuel). For this study, we have used empirically determined vegetation mass values at various locations on Site and assumed that they represent available fuel.

Table 2-3 summarizes fuel loading values from Site vegetation. Sestak and Riebau (1988) present values for the proportion of available fuel that will be consumed in a fire. For grass, they estimate that 90% will be consumed. For sagebrush, the proportion is 70% and for wood, 50 percent. In general, most wildfires at the Site would consume primarily grass.

Table 2-3. Biomass Values for Fuel Loading Calculations

Community	Biomass Type	1993-1994 Mean (g/m^2)
Xeric	Current year production	126.4
	Litter	189.4
Mesic	Current year production	118.8
	Litter	191.1
Reclaimed	Current year production	129.7
	Litter	189.0

Notes:
g/m^2 =grams per square meter

2.4.4 Fire Duration

For a prescribed burn, the fire duration can be calculated based on the burn plan and the target meteorological conditions. For a wildfire scenario, the burn duration must be estimated based on the area assumed to burn, characteristics of the fire itself, and weather conditions. For preparation of future wildfire scenarios, the burn duration will be determined through discussions with the Colorado State Forest Service or through the use of fire behavior models developed by federal land management agencies (e.g., BEHAVE [Andrews and Bevins, 1998]).

2.4.5 Actinide Emissions

Actinide emissions from a wildfire are a function of the mass of particulate released during the fire, the amount of contaminated soil on the vegetation that is burned, the activity in that soil and, depending on isotope, the amount of actinide taken up by vegetation. The particulate emission estimation methods discussed above represent emissions of vegetation residue. A small amount of attached soil is assumed to be associated with the vegetation residue in the same proportion that it is associated with unburned vegetation. For more readily translocated isotopes, a portion of the activity in the root zone soil is also assumed to be incorporated in plant tissue and, therefore, in the residue released during combustion. Soil contamination levels in the area burned may then be used to convert mass emissions to activity units for various isotopes of interest, subject to the variables described below.

Mass Loading—How much soil is attached to vegetation surfaces? Pinder, et al. (1989) summarized the results of their own studies and those of other researchers and reported that soil attachment ranges from 1.4 milligrams (mg) soil per g vegetation to 250 mg/g (dry weight basis). Soil attachment varies with plant type—broadleaved plants carry more soil than narrow-leaved plants, while shorter plants carry more soil per unit weight than taller plants, which include the weight of cleaner, upper portions of the

vegetation. An average value obtained from measurements on and near the Site is reported by Arthur and Alldredge (1982) as 18 mg/g and probably represents a reasonable single value for Site vegetation.

Dilution of Activity— As discussed in Section 2.2.1, measurements at the Site support dilution of soil on vegetation relative to the underlying soil activity. This issue has also been explored conceptually in the development of DOE's RESRAD risk assessment model (Chang, et al., 1998; Gilbert, et al., 1983) and has been empirically determined in a few studies. Generally the RESRAD developers and their colleagues have assumed that the mass loading of particulates in air downwind from a radiologically contaminated soil area will include both contaminated particles originating from the finite contaminated area and "clean" particles originating from an infinite surrounding area of uncontaminated soil. The proportion of contaminated to clean particles will decrease in proportion to the size of the contaminated area and with distance downwind from the contamination. On a theoretical basis, they have concluded that directly above the contaminated area, the proportion of contaminated particles to clean would range from negligible to perhaps a 50/50 mix (Gilbert et al., 1983). At most points downwind, the proportion should be less.

Based on the above data and inference, use of the underlying soil activity to represent the activity of soil on vegetation surfaces is probably a reasonable upper bound across the Site. Use of a diluted activity, say 20%, may be warranted if use of the conservative upper-bound values appears to yield unreasonably conservative overestimates of emissions.

Plant Uptake—As noted above, the uptake of actinides by plants may be significant for some isotopes other than Pu and Am. A transfer coefficient approach is used in risk assessment models to account for this factor, where the transfer coefficient expresses the elemental concentration of a given isotope in vegetation dry weight relative to the elemental concentration of the isotope in the root zone soil (dry weight). Baes, et al. (1984) give average transfer coefficients for both vegetative and reproductive plant tissues for elements of interest:

- Plutonium isotopes: 4.5×10^{-4} (vegetative); 4.5×10^{-5} (reproductive);
- Americium isotopes: 0.0055 (vegetative); 2.5×10^{-4} (reproductive); and
- Uranium isotopes: 0.0085 (vegetative); 0.004 (reproductive).

How do these values compare with the activity in soil attached to vegetation surfaces? Arthur and Alldredge (1982) found an overall mean value of 15.0 pCi/g for Pu attached to plant surfaces at the Site, compared to a mean Pu concentration of 0.70 pCi/g for washed vegetation samples. Later investigations by Webb et al., 1994, showed that washing did not always remove all the soil attached to vegetation surfaces; therefore, the 0.70 pCi/g value probably overstates the concentration of Pu in plant tissue. These values indicate

that, at least for Pu, soil attachment is a more significant source of Pu from burning vegetation than plant uptake.

Given the large range of soil mass loading values seen in the literature and the uncertainty of the dilution effect, the additional activity due to root uptake and translocation may or may not be significant for a given isotope. If desired, the transfer coefficients given by Baes, et al. (1984) may be factored into the conversion of particulate emissions during a fire to actinide emissions.

2.5 Emission Estimation Methods for Anthropogenic Activities

Methods for estimating particulate and actinide emissions from anthropogenic activities such as remedial action, building demolition, and prescribed burning are identified in this section. Implementation of these methods will occur in the continuation to this study. The EPA has established predictive equations for estimating fugitive dust emissions from these types of activities, which are documented in *Compilation of Air Pollutant Emission Factors, AP-42* (EPA, 1995a). The equations in AP-42 reflect the guidance also presented in *Models for Estimating Air Emission Rates from Superfund Remedial Actions* (EPA, 1993) and *Control of Open Fugitive Dust Sources* (EPA, 1988).

Depending on the scenario evaluated, it is anticipated that some set of the following equations will need to be employed. The basic equations and inputs from the AP-42 guidance have been reiterated here. Where relevant, the appropriate EPA guidance will be used to fully apply any given equation. Furthermore, the guidance provides information related to the effectiveness of various control techniques. This information will also be used in developing emission estimates.

2.5.1 Remediation Activities

Remediation activities on Site will involve particulate emission estimation for various material transfer operations, such as excavating, adding to or removing soil from piles, truck dumping, and soil grading. It is expected that emissions from vehicular traffic on paved and unpaved roads will also need to be estimated. The equations identified in the EPA guidance are proposed for future use in the actinide migration study for the Site, and are defined below. The text reflects excerpts from the AP-42 guidance. Actinide emissions will be determined based on particulate emissions and the actinide concentration of the material emitted (i.e., pCi/g for whatever isotopes are appropriate).

Unpaved Roads— An empirical expression used to estimate the quantity in pounds (lb) of size-specific particulate emissions from an unpaved road, per vehicle mile traveled (VMT), is:

$$E = \frac{k\left(\frac{s}{12}\right)^{a}\left(\frac{W}{3}\right)^{b}}{\left(\frac{M}{0.2}\right)^{c}}$$

where:

k, a, b and c are empirical constants;
E is the size-specific emission factor (lb/VMT);
s is the surface material silt content (%);
W is the mean vehicle weight (tons); and
M is the surface material moisture content (%).

The source characteristics W, s, and M are referred to as correction parameters for adjusting the emission estimates to local conditions. The constants, based on the stated aerodynamic particle sizes, are shown in Table 2-4. The above equation was developed from tests of traffic on unpaved surfaces, either uncontrolled or watered. The ranges of source conditions that were tested in developing the equation are shown in Table 2-5. Appendices C.1 and C.2 of AP-42 contain field and laboratory procedures for determining road surface silt and moisture content. (The unpaved road equation is routinely used at the Site in calculating particulate emissions from new activities. As a result, Site-specific data have been established or assumptions have been made for the parameters noted above.)

Table 2-4. Constants for Unpaved Roads Equation

Constant	$PM_{2.5}$	PM_{10}	PM_{30}[a]
k (lb/VMT)	0.38	2.6	10
A	0.8	0.8	0.8
B	0.4	0.4	0.5
C	0.3	0.3	0.4

[a]Assumed equivalent to total suspended particulate

Notes:
lb/VMT = pounds per vehicle mile traveled

Table 2-5. Range of Source Conditions for Application of Unpaved Roads Equation

Surface Silt Content (%)	Mean Vehicle Weight		Mean Vehicle Speed		Mean No. of Wheels	Surface Moisture Content (%)
	Mg	ton	km/hr	mph		
1.2 to 35	1.4 to 260	1.5 to 290	8 to 88	5 to 55	4 to 7	0.03 to 20

Notes:
Mg = megagrams (1,000 grams)
km/hr = kilometers per hour
mph = miles per hour

Paved Roads— An empirical expression used to estimate the quantity of size-specific particulate emissions from a paved road is:

$$E = k\left(\frac{sL}{2}\right)^{0.65}\left(\frac{W}{3}\right)^{1.5}$$

where:

E is the particulate emission factor (having units matching the units of k);
k is the base emission factor for particle size range and units of interest (see Table 2-6);
sL is the road surface silt loading (g/m^2); and
W is the average weight (tons) of the vehicles traveling the road.

The particle size multiplier (k) varies with aerodynamic size range, as shown in Table 2-6. As with the unpaved road equation presented previously, this equation is routinely used at the Site to estimate particulate emissions.

Emissions from paved roads may increase due to the carryout of mud and dirt (EPA, 1988). Emission factors have been developed based on surface loading measurements, and are stated in terms of grams per vehicle pass. The emissions estimated by these factors, shown in Table 2-7, represent particulate (<10 μm) generated over and above the "background" for the paved road in question.

Storage Piles— Dust emissions from storage piles occur at several points in the storage cycle, such as material loading onto the pile, disturbances by strong winds, and loadout from the pile. The quantity of dust emissions from aggregate (sand, gravel, soil, etc.) storage operations varies with the volume of aggregate passing through the storage cycle. Emissions also depend on three parameters that describe the condition of a particular storage pile: age of the pile, moisture content, and proportion of aggregate fines.

Table 2-6. Particle Size Multipliers for Paved Roads Equation

Size Range[a]	Multiplier k[b]		
	g/VKT	g/VMT	lb/VMT
$PM_{2.5}$[c]	1.1	1.8	0.004
PM_{10}	4.6	7.3	0.016
PM_{15}[d]	5.5	9.0	0.020
PM_{30}	24	38	0.082

[a]Refers to airborne particulate matter (PM_x) with an aerodynamic diameter equal to or less than x micrometers.
[b]The multiplier k includes unit conversions.
[c]Ratio of $PM_{2.5}$ to PM_{10}.
[d]Often used as a surrogate for total suspended particulate.

Notes:
g/VKT = grams per vehicle kilometer traveled
g/VMT = grams per vehicle mile traveled
lb/VMT= pounds per vehicle mile traveled

Table 2-7. Factors for Mud/Dirt Carryout From Paved Roads

Particle Size Fraction	Sites With > 25 Vehicles/Day[a]	Sites With < 25 Vehicles/Day[a]
< about 30 μm	52	19
< 10 μm	13	5.5
< 2.5 μm	5.1	2.2

[a]Factors expressed in g/vehicle pass. Mean values of factors are shown.

When aggregate is loaded onto a storage pile, the potential for dust emissions is at a maximum. Fines are easily separated and released to the atmosphere upon exposure to air currents, either from aggregate transfer itself or from high winds. As the aggregate pile weathers, however, the potential for dust emissions is greatly reduced. Moisture causes aggregation and cementation of fines to the surfaces of larger particles. Any significant rainfall soaks the interior of the pile, and the drying process is then very slow.

Adding or removing aggregate material to/from a storage pile usually involves dropping the material onto a receiving surface. Truck dumping on the pile or loading out from the pile to a truck with a front-end loader are examples of batch drop operations. Adding material to the pile by a conveyor stacker is an example of a continuous drop operation.

The quantity of particulate emissions generated by either type of drop operation, per kg of material transferred, may be estimated using the following empirical expression:

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$$E = k \frac{(0.0016)\left(\frac{U}{2.2}\right)^{1.3}}{\left(\frac{M}{2}\right)^{1.4}}$$

where:

E is the emission factor (kg/megagram [Mg]);
k is the particle size multiplier (dimensionless);
U is the mean wind speed, meters per second (m/s); and
M is the material moisture content (%).

The particle size multiplier in the equation, k, varies with aerodynamic particle size range, as shown in Table 2-8. The factor for < 50 μm is 1.0.

Table 2-8. Particle Size Multipliers For Storage Pile Equation

Aerodynamic Particle Size Multiplier (k) For Storage Pile Equation				
< 30 μm	< 15 μm	< 10 μm	< 5 μm	< 2.5 μm
0.74	0.48	0.35	0.20	0.11

Total suspended particulates from wind erosion of continuously active piles can be estimated as:

$$E = 0.19\left(\frac{s}{1.5}\right)\left[\frac{(365-p)}{235}\right]\left(\frac{f}{15}\right)$$

where:

E is the emission factor ($g/m^2/day$);
0.19 is an empirical constant ($g/m^2/day$);
s is the percentage silt of aggregate (%);
365 is the number of days per year;
p is the number of days of precipitation > 0.01 inch per year;
235 is an empirical constant (days);
f is the fraction of time the wind is > 5.4 m/s at mean pile height (unitless); and
15 is an empirical constant (unitless).

Excavation— An equation typically used for estimating total suspended particulate (less than or equal to 30 μm) emissions from excavation is:

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$$E = \frac{0.0046\,(d)^{1.1}}{(M)^{0.3}}$$

where:

E is the particulate emission factor (kg of dust per m^3 of soil excavated);
d is the drop height (m); and
M is the material moisture content (%).

Grading— The equation for total suspended particulate (less than or equal to 30 μm) emissions from surface grading operations is:

$$E = 0.0034\,(S)^{2.5}$$

where:

E is the particulate emission rate (kg/VKT);
VKT is the vehicle kilometers traveled; and
S is the mean vehicle speed (km/hr).

Wind Erosion of Disturbed Surfaces—Wind erosion from level areas of soil that are exposed during remediation may be estimated using the wind erosion models identified in Section 2.3 of this report. Also, measurements of the dust resuspension for disturbed soil conditions were made during the wind tunnel experiments conducted at OU3 in 1993. Alternatively, estimates of the dust resuspension flux could be obtained from an empirical relationship derived from this data set. The length of time it takes a disturbed area to recover, so that "undisturbed" emission factors again apply, will depend on how quickly the area is revegetated, and on the tendency for the soil to promote crust formation. Clay content in soil, for example, tends to promote crust formation.

2.5.2 Demolition Activities

Operations in the demolition and removal of structures from a site include mechanical dismemberment, debris loading, on-site truck traffic, and bulldozing. No emission factor exists for wrecking a building. For this operation, EPA guidance suggests use of the materials handling equation identified for storage piles in the previous subsection. For on-site truck traffic, the equations developed for paved and unpaved roads are recommended.

Debris Loading— For debris loading, an emission factor based on the filling of trucks with crushed limestone, using a front-end loader, has been determined. The emission factor is:

$$E = k(0.01305)$$

where:

E is the particulate emission factor (kg/m^2);
k is the particle size multiplier (see Table 2-8); and
0.01305 incorporates an average measured total suspended particulate emission factor.

Bulldozing— The emission factor for bulldozing operations comes from measurements of overburden removal at western surface coal mines. The AP-42 equation for total suspended particulate emissions (< 30 μm) is:

$$E = \frac{2.6(s)^{1.2}}{(M)^{1.3}}$$

where:

E is the particulate emission rate (kg/hr);
s is the silt content of the surface material (%); and
M is the moisture content of the surface material (%).

2.5.3 Prescribed Burns

Prescribed burns at the Site may release actinides just as a wildfire would. The method that will be used to calculate emissions from a prescribed burn is the same as that described previously for estimating wildfire emissions. The main difference between estimating emissions for prescribed burns and wildfires is that some of the parameters that affect emissions will be preset for a prescribed burn but would be highly variable for a wildfire. These parameters include the size and location of the burn, the burn duration, and other elements of the burn plan, which affect the intensity, direction of spread, and rate of spread of the fire.

2.6 Emission Estimation Conclusions

Actinide resuspension at the Site is an ongoing phenomenon that is episodic in nature and influenced primarily by meteorological variables (wind speed and rainfall); particle and soil properties (moisture level and particle density); and surface characteristics (density and type of vegetative growth, and snow cover). Given the density of vegetation within the contaminated soil areas on Site, direct resuspension of actinides from exposed soil surfaces is considered insignificant (assuming no disturbance of the area). The source of contaminated soil resuspension is, then, the dust-laden vegetation and litter.

A significant amount of research in particle and actinide resuspension has occurred over the years. This research emphasizes the need to customize any approach to the particular location of interest. The unique meteorological, soil, and surface characteristics must be taken into account to produce a reliable emission estimation approach for a given area.

Past wind tunnel experiments on Site relate dust resuspension to ambient wind speed and currently provide the best method for estimating emissions. Site wind tunnel data indicate dust resuspension varies with wind speed raised to the third power.

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3.0 DISPERSION AND DEPOSITION MODELING

This section discusses the methodology followed in conducting the air dispersion and deposition modeling analyses, including model selection, model input data, and model results.

3.1 Model Selection

As discussed in Section 1.0, the purpose of this task was to select and implement a dispersion and deposition model to track actinide migration at the Site through the air pathway. Because the model will be used to estimate actinide dispersion and deposition from a variety of land use, remediation, and D&D scenarios, it must simulate dispersion of actinide-laden particles from a variety of sources at the Site and calculate both concentrations in air and deposition of particles to water or ground surfaces. Required model features included the capability to:

- Simulate dispersion from multiple source types (i.e., point, area, volume sources);
- Predict annual deposition and air concentrations in the near-field (within 0 to 5 km of a particular emission source or of the Site);
- Use relatively limited meteorological input data (i.e., data available from on-Site tower or other nearby stations); and,
- Accept varying emission rates.

While conversion of air concentrations to dose units was also desired, it was not a significant factor in choosing the appropriate model. The ability to calculate regional-scale deposition/concentration also was not needed.

Several models are available to conduct air concentration and/or radioactive dose assessments for the Site. These include models designed specifically to determine compliance with federal regulations concerning air quality and public dose standards, and those designed for emergency response and risk assessment. Most of the regulatory compliance models have been developed and maintained by EPA. In addition, Argonne National Laboratory, Lawrence Livermore National Laboratory, and others have developed specialized risk assessment and emergency response models for DOE facilities. A cursory evaluation of the available models was conducted against the criteria outlined above.

3.1.1 Model Evaluation

This section presents an overview of the types of models considered for this study and discusses in somewhat more detail the characteristics of the chosen model.

EPA Regulatory Air Quality Models—The EPA Office of Air Quality Planning and Standards (OAQPS) Support Center for Regulatory Air Models Internet website (**http://www.epa.gov/scram001**) provides guidance and information on atmospheric dispersion (air quality) models that support regulatory programs required by the federal Clean Air Act. Numerous models are available to evaluate special meteorological and source release situations. However, models designed primarily for screening analyses, regional-scale applications, or complex atmospheric or plume chemistry simulations were generally considered unsuitable for the current application. Given the averaging times and spatial resolution needed in the results from this task, only a subset of the regulatory air models were considered for the actinide migration study.

Because of its flexibility in determining concentration or deposition in flat or simple terrain, the primary regulatory model used in evaluating compliance with federal air quality standards (and the candidate model considered for this study) is the Industrial Source Complex (ISC3) model. ISC3 includes algorithms for calculating both short-term (ISCST3) and long-term (i.e., annual, ISCLT3) concentration or deposition, although the names are somewhat misleading because the short-term model can also be used to calculate annual averages. The latest version of the short-term model, at the time of this study, was ISCST3 Version 98356.

The ISCST3 model was developed and is supported by EPA to predict air concentrations and deposition from multiple source types (i.e., stacks, areas of fugitive emissions, equipment operation, open pits, etc.), a key feature for simulating the types of activities with which the actinide migration study is most concerned. ISCST3 is a Gaussian plume model, where plume spread in both the vertical and horizontal dimensions can be represented by a Gaussian (i.e., bell-shaped) distribution of pollutant mass. The plume spread depends on distance downwind from the release area (source) and various meteorological factors such as wind speed and stability. The time resolution in the short-term model is one-hour and averaging periods up to several years can be handled. The model is suitable for calculating concentration or deposition in simple to somewhat complex terrain at distances up to 50 km from the source.

Recent upgrades to the model have incorporated enhanced dry deposition algorithms and area source modeling capabilities. Because the results of the current air pathway study will be used as input to other actinide migration pathway models (such as erosion and surface water transport), the ability to calculate deposition was important in the model chosen for this study. The enhanced ISCST3 dry deposition model, which handles the gravitational settling and removal of particulates by dry deposition, is based on a dry deposition algorithm (Pleim et al., 1984) contained in the Acid Deposition and Oxidant Model (ADOM). This algorithm was selected as a result of an independent model evaluation study performed by EPA (EPA, 1994).

For the representation of fugitive area sources, the latest version of the ISCST3 model allows for the depiction of irregularly-shaped areas as multi-sided polygons with up to 20 vertices. Older versions of the model required that area sources be represented by simple geometric shapes such as squares or rectangles. This enhancement allowed a more

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realistic depiction of dispersion from differentially contaminated surface soil areas than other candidate models. In addition, the ISCST3 model can calculate concentrations at the boundary of, or even within, larger area sources, which was also important for this study.

Gaussian plume models, in general, offer a trade-off between precision and ease of use. The model inputs, including meteorological data, are generally available and the model set up can be easily modified to simulate different scenarios. However, the results may be considered to have an uncertainty that ranges from a factor of 2 to over an order of magnitude (EPA, 1995a). Longer-term averages (i.e., annual) have less uncertainty than short-term results (i.e., 1 hour) and patterns of concentration/deposition are more reliable than a single-point prediction.

Regulatory Dose Assessment Models—EPA's Radiation Protection Division provides the methods and scientific basis for EPA's radiation exposure, dose, and risk assessments. These assessments, in turn, support the development of EPA policy, guidance, and rulemakings concerning radiation protection and risk management. Among other functions, the Radiation Protection Program develops radionuclide fate and transport models, dose and risk models, and dose and risk coefficients.

A selection of the radioactive dose assessment models available from the Radiation Protection Program **(http://www.epa.gov/radiation/assessment)** is shown in Table 3-1.

Table 3-1. Radioactive Dose Assessment Models

Model Name	Description
PRESTO	Multimedia model for assessing low-level and low activity wastes, Naturally Occurring and Accelerator-Produced Radioactive Materials (NARM), and uranium mill tailings waste.
CAP88-PC	Atmospheric transport model for assessing dose and risk from radioactive air emissions. Applicable to DOE compliance with Clean Air Act.
COMPLY	Atmospheric screening model for assessing dose from radioactive air emissions. Applicable to DOE compliance with Clean Air Act, and Nuclear Regulatory Commission regulations.
Environmental Dosimetry Upgrade Project	Project initiated to develop an updated computer modeling system for evaluating atmospheric releases of radionuclides. This modeling system is an adaptation of the GENII Environmental Radiation Dosimetry System (Napier et al., 1988). The modeling system includes atmospheric transport, terrestrial transport, and dose and risk coefficients which are improvements over the CAP88 computer model.

Results from CAP88-PC and other regulatory dose assessment models are routinely used in annual regulatory compliance reports and environmental assessment studies at the Site and at other DOE facilities. However, these models generally include greatly simplified dispersion algorithms and lack the latest refinements in deposition and area source algorithms that have been implemented in ISCST3. They are unsuitable for estimating actinide deposition as desired by the AME Group.

DOE Planning/Risk Assessment Models—The Residual Radioactive material model (RESRAD), developed by Argonne National Laboratory, is a planning model designated by DOE Order 5400.5 (DOE, 1993) for the evaluation of radioactively contaminated sites. The Nuclear Regulatory Commission (NRC) has approved the use of RESRAD for dose evaluation by licensees involved in decommissioning, NRC staff evaluation of waste disposal requests, and dose evaluation of sites being reviewed by NRC staff. RESRAD is a refined multimedia model designed to calculate site-specific residual radioactive material guidelines as well as radiation dose and excess lifetime cancer risk to a chronically exposed on-site resident. Radiation doses, health risks, soil guidelines, and media concentrations are calculated over user-specified time intervals. The source is adjusted over time to account for radioactive decay and ingrowth, leaching, erosion, and mixing. While RESRAD contains extensive code to calculate radioactive dose and risk, the dispersion algorithms are extremely simple and cannot be used to develop the spatially resolved concentration/deposition data required for the current application

Emergency Response Models—In addition to the above-mentioned dose/risk assessment models, other models have been developed to assess air concentrations and/or radioactive dose from unplanned releases of radioactive material. The most pertinent examples of these types of models for the current study are: the Computer Assisted Protective Action Recommendation System (CAPARS) developed by AlphaTRAC Inc; and, the Atmospheric Release Advisory Capability (ARAC) model developed by Lawrence Livermore National Laboratory.

CAPARS is an emergency response model designed to track hazardous material releases and provide critical information to emergency response personnel. It is an improved version of the original Terrain-Responsive Atmospheric Code (TRAC) model designed for RFETS over a decade ago. The system was developed through a partnership between the DOE's Rocky Flats Field Office and the non-profit Regional Atmospheric Response Center (RARC) in Westminster, Colorado.

ARAC is an emergency response model that uses 3-D transport and diffusion models for real-time assessment of incidents involving nuclear, chemical, biological, or natural hazardous material, and to predict the extent of the spread. The modeling domain can be selected anywhere in the world and scaled to the size of the problem. Wind observations are interpolated and adjusted over the domain to produce a mass-consistent flow in the topographic setting, using a terrain database that covers most of the world at 0.5-km resolution. Online databases support the calculation of time-varying release rates, source geometries, and plume rise for explosion, fire, vent, and spill release mechanisms. Releases of hazardous material are simulated using thousands of "marker particles," each carrying the unique properties of its released material. Multiple sources of nuclear or chemical material can be simultaneously treated. These sources are transported and dispersed in the atmosphere and deposited to the ground using a Lagrangian Monte Carlo diffusion method, which has been validated against numerous tracer studies, including those for the Site. A dose-factor database contains dose-conversion factors for internal

and external exposure to all radionuclides. A toxicological database includes Emergency Response Planning Guidelines or equivalents for hundreds of chemicals.

These models were not selected because they require more refined meteorological input than ISCST3 to characterize the complex wind flows in the region. They are designed to calculate concentration and/or dose at populated areas several kilometers distant to the Site (i.e., the metropolitan areas of Boulder and Denver), rather than the near-field area of most concern for this study.

3.1.2 Model Comparison Studies

Studies have been conducted to compare ISC results at the Site against other available models (Ciolek and Magtutu, 1998; Rood and Till, 1997). These studies, though conducted using an older version of ISC (ISCST2), have nonetheless shown ISC to perform well in determining maximum concentrations. However, these studies compared concentration estimates only (not deposition), failed to provide a comparison of model performance for different source types (i.e, fugitive area releases versus elevated stack releases), and compared concentrations significantly downwind (8 and 16 km) from the Site, rather than results in the near field that is of concern for this study. Note that the latest version of ISC at the time of this study, ISCST3 Version 98356, provides improvements in deposition and area source algorithms that were not available in the tested model, ISCST2.

3.1.3 Selected Model

Because the initial phase of this study called for the modeling of concentration/dose and dry deposition from irregularly-shaped fugitive area sources, the ISCST3 model was considered the best model for the application. With the ability to enter multi-sided polygons, ISCST3 can closely approximate the actual shapes of the area sources at the Site. Therefore, ISCST3 Version 98356 was used to perform refined dispersion and deposition modeling for the air pathway study. ISCST3 is described in detail in the *User's Guide for the Industrial Source Complex (ISC3) Dispersion Models, Volumes I and II* (EPA, 1995b).

3.2 Model Input Data

The ISCST3 model requires the input of detailed source characteristics, meteorological data, dispersion modeling option selections, and locations of modeling receptors. The inputs used for the actinide migration study modeling are described below.

3.2.1 Model Options

The ISCST3 model was used to estimate the transport of airborne actinides from source areas at the Site to the fenceline of the Site (*dispersion*) and the removal of actinide-contaminated particles from the air to soil or water surfaces on or surrounding the Site (*deposition*). Particulates are brought down to the surface through the combined

processes of turbulent diffusion and gravitational settling. Once near the surface, they may be removed from the atmosphere and deposited on the surface.

Dispersion was estimated by using the option within the ISCST3 model that produces output of airborne activity concentration in units of picocuries per cubic meter of air (pCi/m^3), while deposition on ground or water surfaces was estimated in units of picocuries per square meter per year ($pCi/m^2/yr$). From the estimated concentrations, airborne effective dose equivalent in millirems (mrem) was computed by applying conversion factors (described below). Rural dispersion parameters were used for both dispersion/concentration and deposition model runs. For the deposition model runs, the option for plume depletion due to dry removal mechanisms was not included in the model calculations due to computational limitations; only the dry deposition fluxes themselves were calculated. Wet deposition was not modeled with this phase of the study because the contribution to total deposition by wet removal mechanisms was assumed insignificant.

3.2.2 Meteorological Data

An ISCST3 meteorological input file for the Site was created with the use of the EPA Meteorological Processor for Regulatory Models (MPRM). Surface meteorological parameters that were measured at the Site in 1996 were combined with concurrent upper-air and cloud cover data from the National Weather Service (NWS) station in Denver using MPRM. (1996 was chosen because the 1996 Site data set was more complete for certain needed parameters than the 1997 or 1998 data sets.) The output product from MPRM was an hourly meteorological input file that could be used for both concentration and deposition modeling with the ISCST3 model.

Several of the parameters measured at the Site were directly written to the meteorological input file, including 10-m wind speed, wind direction, and surface temperature. Atmospheric stability was computed within MPRM by using the hourly wind speed and the standard deviation of the horizontal wind direction for each hour. The height of the mixing layer above ground for each hour was computed within MPRM from twice-daily mixing height values that were derived from Denver NWS upper-air soundings.

To perform dry deposition calculations, the ISCST3 model requires three additional parameters for each hour. These parameters are roughness length, Monin-Obukhov length, and surface friction velocity. The surface friction velocity is a measure of wind shear stresses at the surface, while the Monin-Obukhov length is a stability parameter that relates this velocity to the transport of heat. Surface roughness is, by definition, the height at which the wind speed diminishes to zero. It is generally proportional to the physical dimensions of the obstructions to the wind flow. A year-round roughness length of 1.5 cm was chosen as representative of the Site. Surface friction velocity and Monin-Obukhov length were computed within MPRM from several parameters, including solar radiation and surface pressure data from the Site, and cloud cover data from the Denver NWS station.

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Other parameters required by MPRM include surface characteristics such as albedo, Bowen ratio (indicator of the amount of moisture at the surface), minimum Monin-Obukhov length, surface heat flux, and anthropogenic heat flux. Values for these parameters were chosen for the Site from MPRM defaults for a rural setting or from seasonal values typical for a grassland area.

Figure 3-1 presents a wind rose for the meteorological input file.

3.2.3 Modeling Receptors

Separate receptor grids were created for deposition modeling and for concentration/dose modeling. For concentration/dose modeling, receptors were placed along the Site fenceline at 100-m receptor spacing. Figure 3-2 shows the receptor grid that was used for the concentration/dose modeling. Deposition modeling was conducted with a receptor grid with 200-m spacing that completely covered the Site and extended at least 1 km beyond the Site fenceline in all directions. In the predominant downwind direction for the area, the grid extended 2 km beyond the fenceline. Figure 3-3 presents the receptor grid that was used for deposition modeling.

Elevations for each modeling receptor were determined from United States Geological Survey (USGS) Digital Elevation Model (DEM) data with the use of ArcInfo® software. A three-dimensional surface that represented the terrain at the Site was created electronically from the DEM data. The two-dimensional receptor grid (without elevations) was then overlaid onto the surface and individual receptor elevations were assigned with a routine within the software.

3.2.4 Source Data

The goal of the initial phase of modeling was to estimate the dispersion and deposition of actinide activity from the resuspension of soil at the Site, focusing on chronic, natural resuspension mechanisms that would be ongoing with or without anthropogenic activity. To accomplish this, Radian had to determine how to best represent the soil surface as sources in the model, and to quantify the amount of actinide activity that would be emitted from the surface with resuspension. Section 2.0 of this report addresses the quantification of actinide emissions from soil resuspension; the representation of the Site emission sources in the ISCST3 model is described below.

To represent the soil surface as emission sources within the ISCST3 model, Radian first obtained maps that provided contours of activity (in pCi/g) for each of the five actinides considered for this study. These maps were developed by Rocky Mountain Remediation Services, L.L.C. (RMRS) from a Site soil sampling database to provide input into annual compliance demonstration modeling required by 40 CFR 61, Subpart H. Radian electronically digitized each activity contour and then used a routine within AutoCAD Map® (Release 2.0) software to reduce each contour to a series of points that created multi-sided polygons that approximated the contour's shape. Other landuse maps for the Site were used to determine the areas within each contour that consist of material that

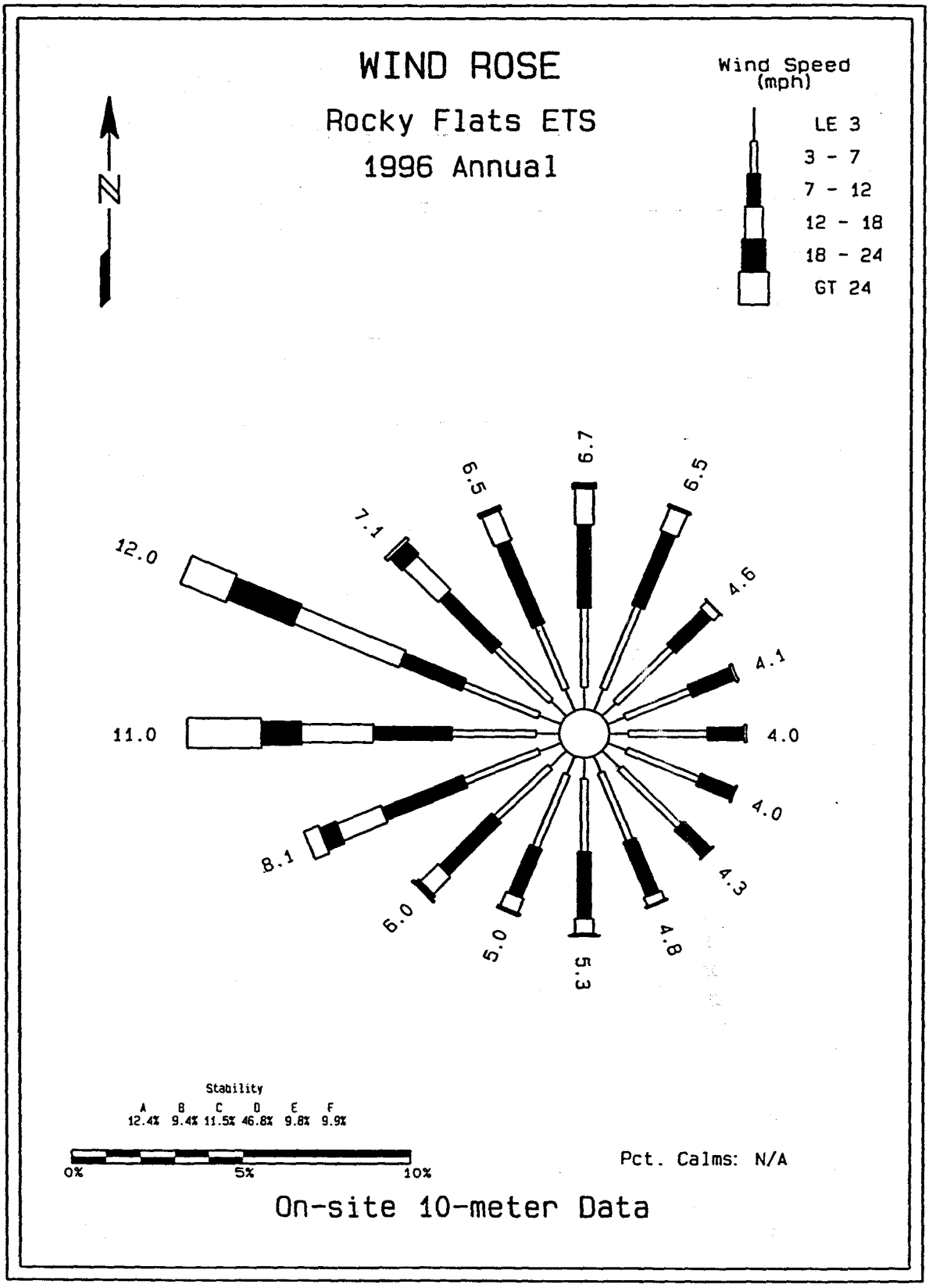

Figure 3-1. 1996 Annual Wind Rose

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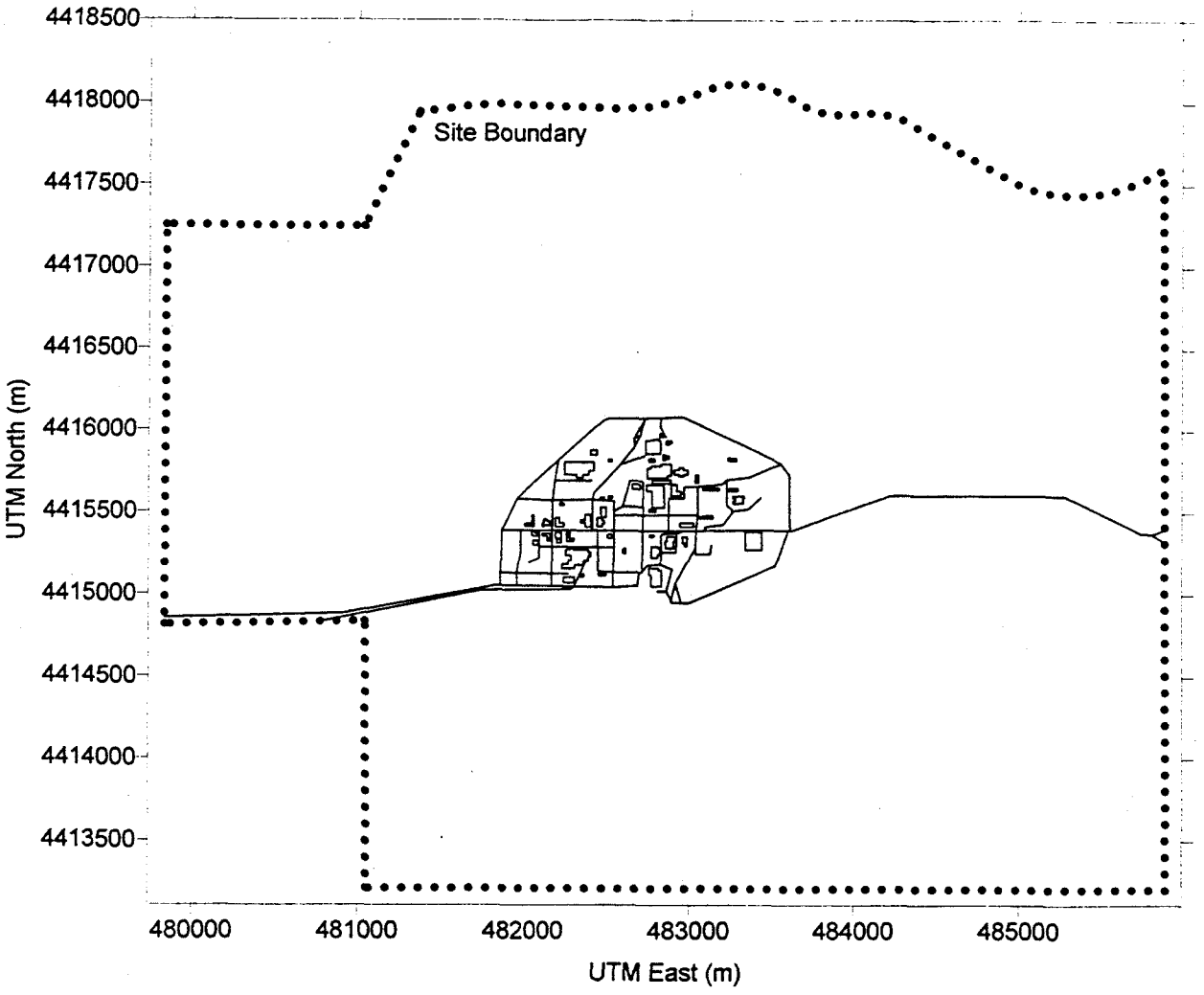

Figure 3-2. Receptor Grid for Concentration/Dose Modeling

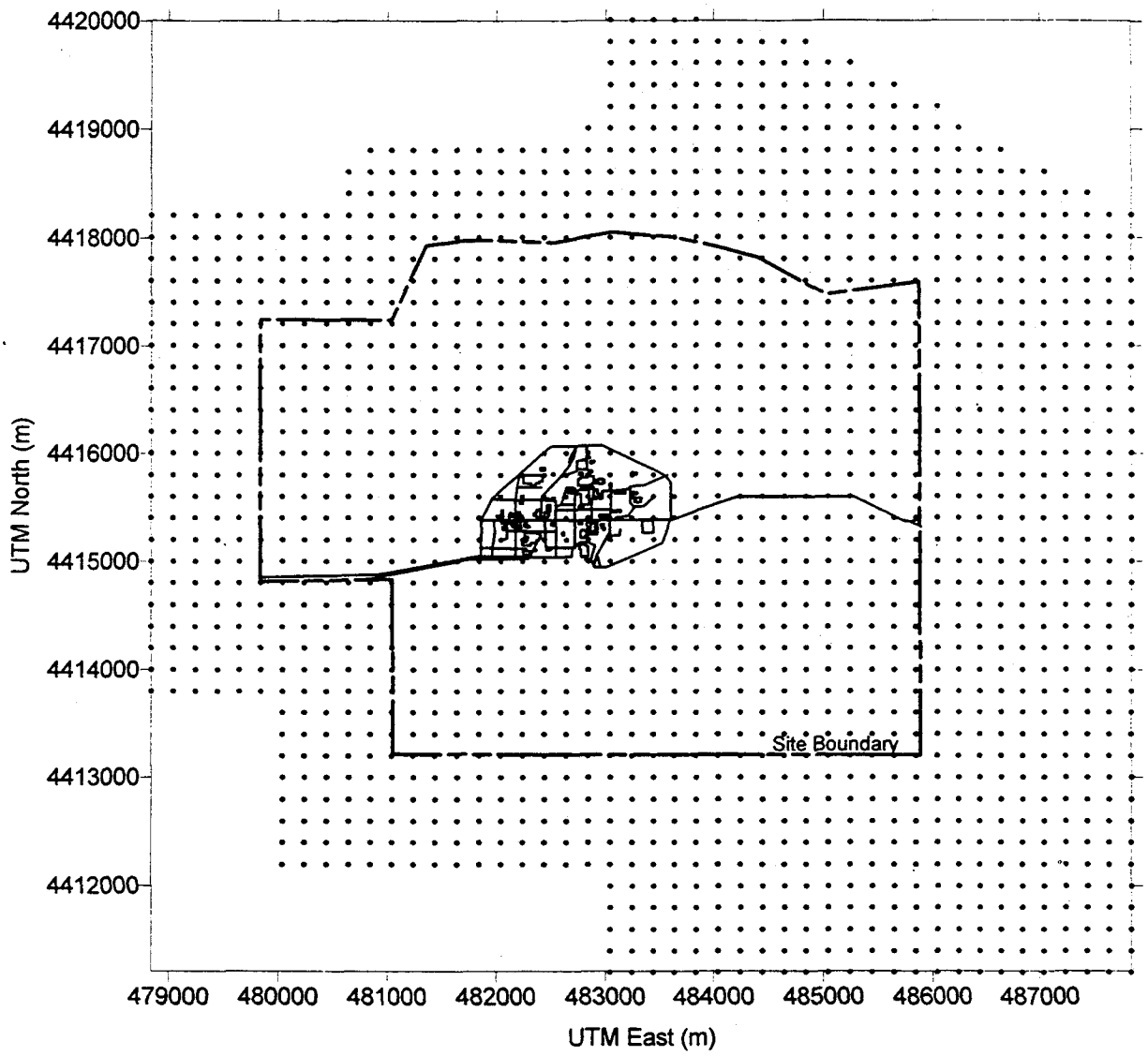

Figure 3-3. Receptor Grid for Deposition Modeling

would not be eroded, specifically the paved areas within the Industrial Area. These nonerodable areas were removed from the area source polygons. The Universal Transverse Mercator (UTM) coordinates for the points that comprised each final contour were then entered into the model.

The activity contours were modeled as area sources, each with release heights at the surface. Emissions for each area source, in units of picocuries per square meter per second ($pCi/m^2/s$), were input to the model with an hourly emissions file. Because the hourly area source emissions associated with this study were quite small, there was a concern that the model would not produce useful output because the display of results in the model output file is limited in precision. Therefore, the area source emissions were scaled by factors of 1,000 to 1,000,000 to guarantee sizeable results. The model output was then divided by these factors to arrive at estimated activity concentration estimates and deposition flux values.

The ISCST3 deposition algorithm requires the input of source variables for settling and removal. These variables (for each source) include particle diameter size categories, corresponding mass fractions (between 0 and 1) for each of the categories, and the corresponding particle density (g/cm^3) for each of the categories. Radian represented resuspended soil at the Site with three particle size categories as follows:

- Category 1: 3 μm diameter (for <3 μm particles), mass fraction =1, particle density = 2.5 g/cm^3;
- Category 2: 10 μm diameter (for 3-15 μm particles), mass fraction =1, particle density = 2.5 g/cm^3; and
- Category 3: 15 μm diameter (for >15 μm particles), mass fraction =1, particle density = 2.5 g/cm^3.

The particle size categories were chosen because of the availability of data on the activity associated with particles in these size categories. These data were used in formulating area source emissions for each particle size and actinide (see Section 2.0). Within the ISCST3 model, the size categories represent the mass-mean particle diameter in units of micrometers for each category. Deposition of activity for each size category was modeled separately by establishing a complete set of area sources for the three sizes of particles. Variations in the area source emissions for the three categories were accounted for within the hourly emission file for each actinide. Because each size category was modeled by itself, rather than in combination with other particle sizes, a mass fraction of one was assigned for each category. The assumed particle density of 2.5 g/cm^3 is considered to be representative of a mineral topsoil (Brady, 1974).

3.2.5 Input Parameters for Fire Scenarios

As discussed in Sections 2.4 and 2.5.3, future use scenarios may include emissions from wildfires and/or prescribed burns. The nature of these events requires somewhat different

model inputs than more routine source types. This section briefly describes the model input parameters that must be estimated for scenarios involving either wildfires or prescribed burns. A more detailed discussion is contained in Appendix A.

Fires will be modeled in future scenarios as one or more area sources in ISCST3, with a release height above the ground of 0.0 m, an initial vertical dimension based on estimated plume rise, and lateral dimensions based on the size of the burned area. The size of the burned area and other factors affecting plume rise would be preset for a prescribed burn but would be more variable for a wildfire scenario. The simulation may include one source area or multiple areas, depending on the overall size of the fire, the burn rate, and the location of the receptors of interest. Actinide and particulate emissions from each area will be based on unit emission estimations calculated as described in Sections 2.4 and 2.5.3.

Wind speed and stability are parameters that vary during the day and year. To calculate plume rise, an average wind speed and stability during the burn will probably be used. For prescribed burns, the target meteorological conditions during which the burn will be conducted will be used to set these parameters. The parameters used for a wildfire will depend on the desired scenario (duration of burn, assumed weather conditions, etc.). It is also possible (but cumbersome) to vary these parameters on an hourly basis (which is the time step used in the meteorological data) using various options available in ISCST3 if a specific burn scenario with detailed resolution is desired.

Once plume rise has been calculated, the initial vertical dimension of the area source or sources that represent the fire can be determined. The *User's Guide for the Industrial Source Complex (ISC3) Dispersion Models, Volume I* (EPA, 1995b) recommends setting the initial vertical dimension to the plume height divided by 2.15 for surface-based sources, such as a fire.

To complete the discussion of modeling fire scenarios, it should be noted that any burn will only be modeled for the hours representing the burn duration and any desired time before or after the fire.

3.3 Dose Conversion

In addition to calculating airborne concentrations of actinides (in units of activity per unit volume of air, e.g., pCi/m^3), results have also been converted to dose units. Effective dose equivalent, measured in units of Sieverts or mrem, represents the amount of radiation energy absorbed per gram of tissue, weighted by its potential to do damage and the susceptibility for harm to different tissues in the human body.

Conversion from activity to effective dose equivalent depends not only on the isotope and the type of radiation it emits, but also on assumptions about exposure pathways and scenarios. To simplify this conversion, we have used conversion factors from EPA air regulations that are based on standard assumptions about exposure pathways and scenarios, as described below.

Regulation 40 CFR 61 contains requirements governing emissions of hazardous air pollutants from certain source types. DOE facilities such as RFETS are subject to the standards of Subpart H, which limits radionuclide emissions from the facility to those amounts that would result in an annual dose to the public of no more than 10 mrem. Appendix E to 40 CFR 61 gives a table (Table 2) of radionuclide concentration values (by isotope) that can be compared to measured radionuclide concentrations in air to demonstrate compliance with the Subpart H standard. If a person was exposed to air containing a given isotope at the concentration levels listed in Table 2 for a full year (under the standard exposure assumptions inherent in these values), they would receive a 10 mrem effective dose equivalent. Therefore, the Table 2 concentration levels can be used to convert between radionuclide concentrations (in curies per cubic meter[Ci/m^3] or pCi/m^3) and effective dose equivalent (in mrem).

For the isotopes of interest in this study, the concentration levels from Appendix E, Table 2 are:

- Am-241 1.9×10^{-15} Ci/m^3
- Pu-239/240 2.0×10^{-15} Ci/m^3
- U-233/234 $7.1/7.7 \times 10^{-15}$ (use 7.7) Ci/m^3
- U-235 7.1×10^{-15} Ci/m^3
- U-238 8.3×10^{-15} Ci/m^3

Each of these isotopic concentrations equates to a 10 mrem/yr effective dose equivalent annual rate for the purposes of this modeling study.

For modeling, emissions in units of activity per unit area per unit time ($pCi/m^2/s$) for a given isotope were input, and the results were then converted to output units of mrem. The conversion factor for each isotope used the previously listed concentration values, plus the appropriate conversions between Ci and pCi and between a 10 mrem and 1 mrem level.

3.4 Modeling Results

This section provides a summary of the results of the ISCST3 modeling for dispersion/concentration and deposition. Each modeling scenario is described briefly, as are the results of each model run. The results of each run are also presented graphically.

3.4.1 Dispersion/Concentration Runs

The dispersion of the five actinides, and the resulting activity concentration at the Site fenceline, was estimated with the ISCST3 model in units of pCi/m^3. Table 3-2 provides the maximum predicted concentrations.

The maximum estimated annual concentration of Pu-239 at the perimeter of the Site was 2.25×10^{-5} pCi/m^3. This maximum was predicted to occur at the eastern fenceline of the Site, directly east of the southern portion of the Site Industrial Area. Figure 3-4 shows

Table 3-2. Results of Concentration Model Runs

Actinide	Maximum Estimated Concentration (pCi/m^3)	Location of Maximum (UTM m East, UTM m North)
Pu-239	2.25×10^{-5}	485878, 4415013
Am-241	3.64×10^{-6}	485877, 4415515
U-233/234	7.82×10^{-9}	485877, 4415515
U-235	1.87×10^{-9}	485817, 4415415
U-238	1.91×10^{-8}	485880, 4414311

Notes:
Am = americium
m = meters
pCi/m^3 = picocuries per cubic meter
Pu = plutonium
U = uranium
UTM = Universal Transverse Mercator

the estimated annual impacts for Pu-239 along the Site fenceline. This location is generally downwind of the surface soil areas contaminated with Pu-239 in the area east and southeast of the 903 Pad. (Appendix B lists the Pu-239 surface soil concentration isopleths used for this study and shows the location of the centroid of each isopleth. Each centroid represents the center of mass of the radionuclide surface soil concentrations within a given isopleth.)

For Am-241, the maximum estimated fenceline concentration was approximately one order of magnitude lower than Pu-239 at 3.64×10^{-6} pCi/m^3. This maximum was predicted to occur approximately 500 m to the north of the maximum for Pu-239, at the Site fenceline directly east of the central portion of the Industrial Area. Figure 3-5 shows the estimated annual impacts for Am-241 along the Site fenceline. This maximum impact area is also generally downwind from the 903 Pad area on an annual basis. Am-241 surface soil concentration isopleth data are shown in Appendix B.

Estimated maximum annual fenceline concentrations for uranium were approximately three orders of magnitude lower than they were for plutonium. For U-233/234, the estimated maximum of 7.82×10^{-9} pCi/m^3 was predicted to occur at the same location as the maximum for Am-241. As shown in Appendix B, however, the primary source area for U-233/234 is near the Solar Ponds, rather than the 903 Pad. Some of the U-233/234 in surface soils in this area is thought to be naturally occurring. Figure 3-6 shows the estimated annual impacts for U-233/234 along the Site fenceline.

Maximum U-235 concentrations were estimated to be 1.87×10^{-9} pCi/m^3, and were predicted to occur approximately 100 m to the south of the maximum for Am-241. The

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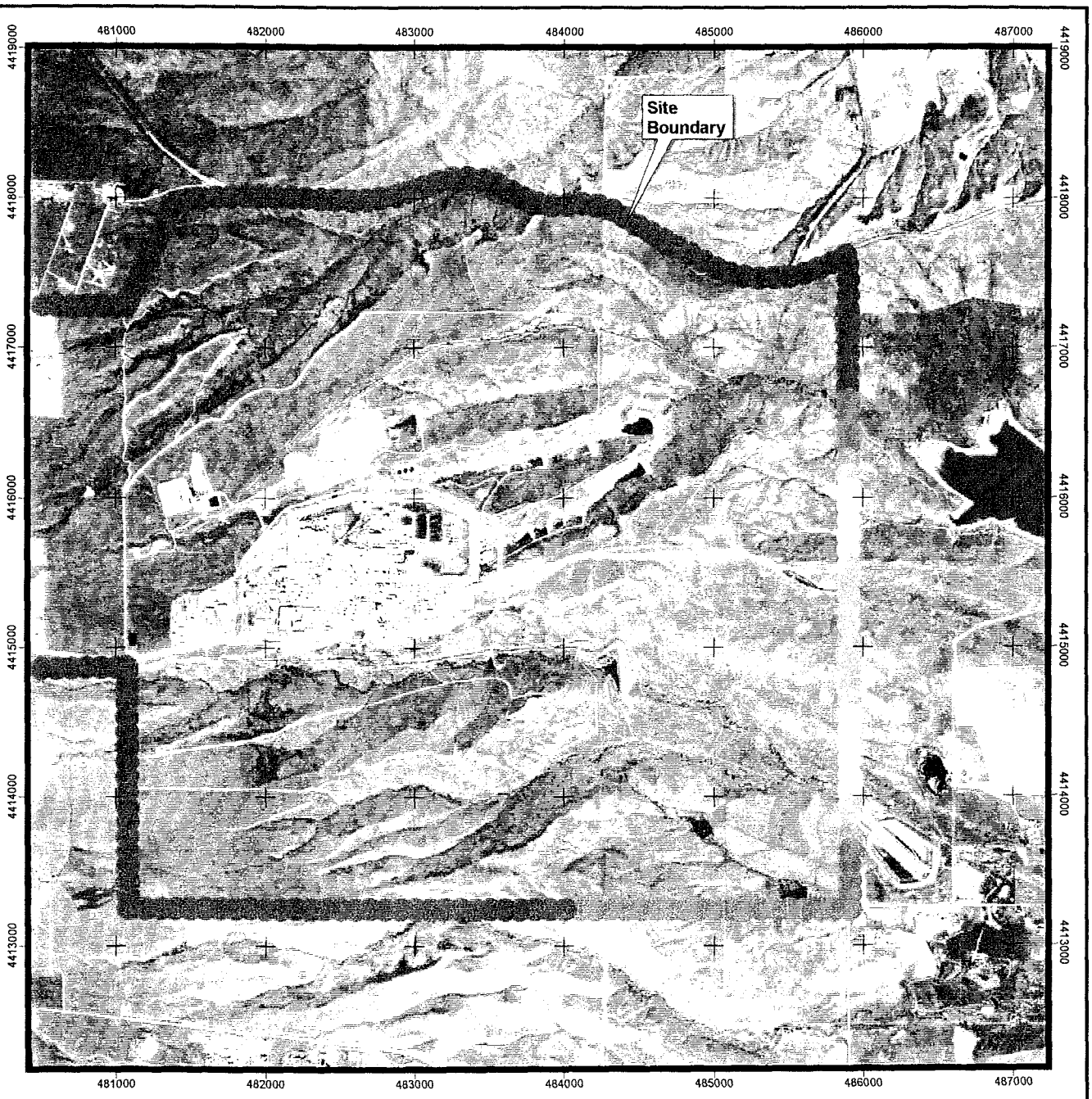

Figure 3-4. Model Predicted Annual Concentration of Activity for Pu-239

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ROCKY FLATS ENVIRONMENTAL TECHNOLOGY SITE
GOLDEN, COLORADO

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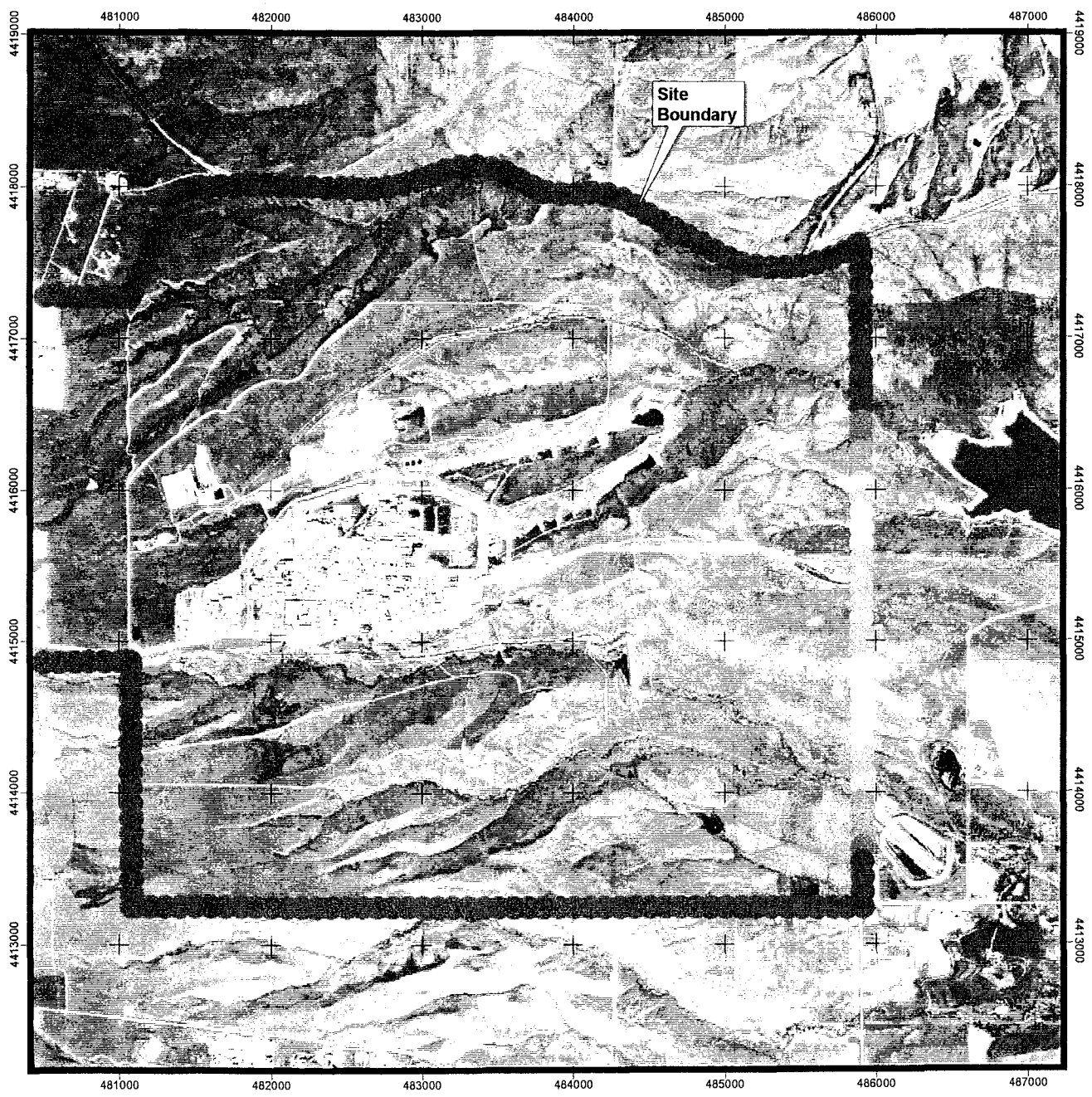

Annual Concentration of Activity
(pCi/m^3) for Am-241

- 1.00E-008 to 3.73E-007
- 3.73E-007 to 7.36E-007
- 7.36E-007 to 1.10E-006
- 1.10E-006 to 1.16E-006
- 1.46E-006 to 1.83E-006
- 1.83E-006 to 2.19E-006
- 2.19E-006 to 2.55E-006
- 2.55E-006 to 2.91E-006
- 2.94E-006 to 3.28E-006
- 3.28E-006 to 3.65E-006

N

0.5 0 0.5 1 1.5 Kilometers

NOTE:
Coordinates reference
UTM Meters, Zone 13, NAD27

Figure 3-5. Model Predicted Annual Concentration of Activity for Am-241

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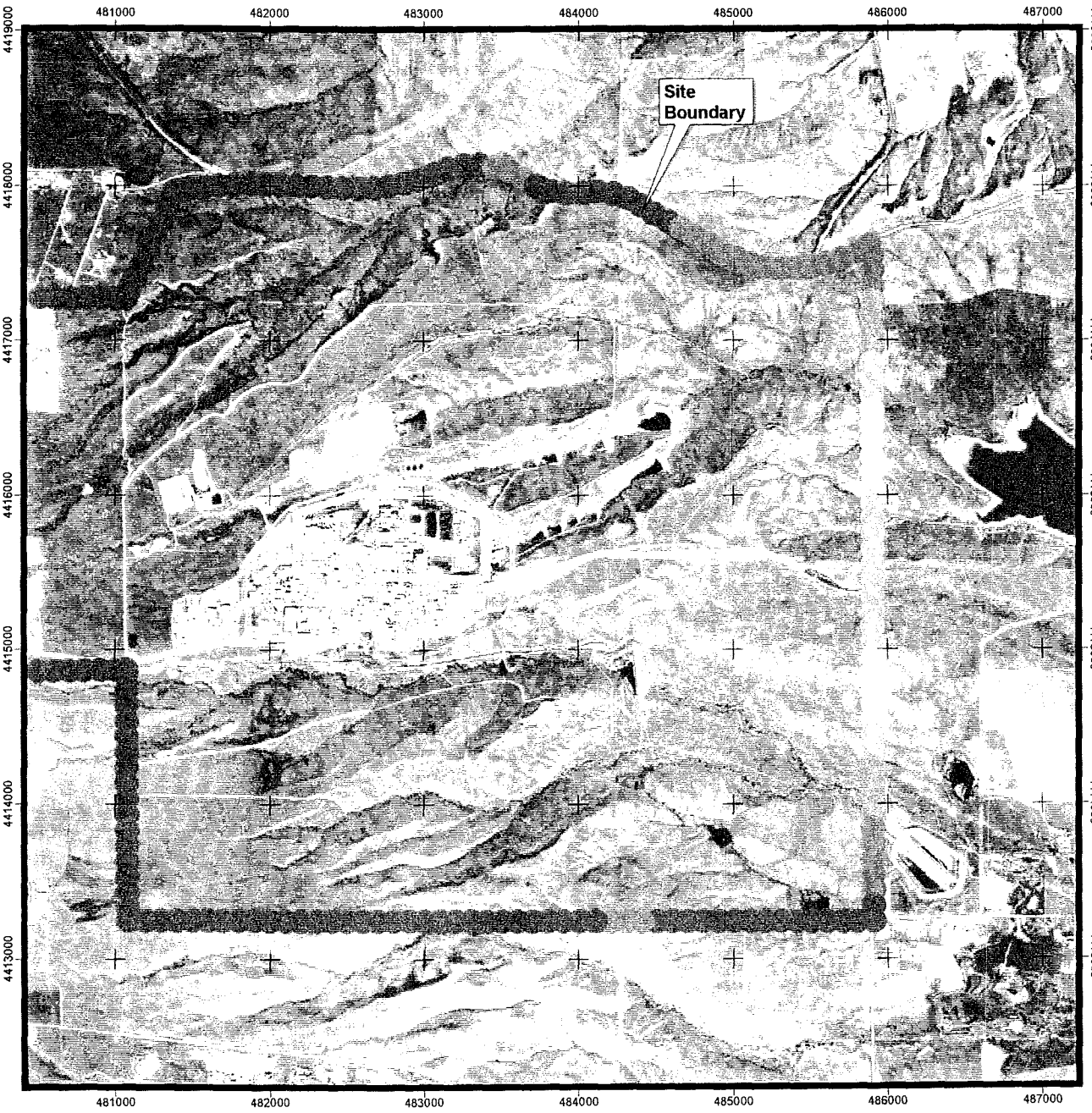

Annual Concentration of Activity
(pCi/m^3) for U-233/234

- 8.00E-011to 8.54E-010
- 8.54E-010 to 1.63E-009
- 1.63E-009 to 2.40E-009
- 2.40E-009 to 3.18E-009
- 3.18E-009 to 3.95E-009
- 3.95E-009 to 4.72E-009
- 4.72E-009 to 5.50E-009
- 5.50E-009 to 6.27E-009
- 6.27E-009 to 7.05E-009
- 7.05E-009 to 7.82E-009

N

0.5 0 0.5 1 1.5 Kilometers

NOTE:
Coordinates reference
UTM Meters, Zone 13, NAD27

Figure 3-6. Model Predicted Annual Concentration of Activity for U-233/234

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major source areas of U-235 are to the west of the Solar Ponds, in the south central portions of the Industrial Area and southwest of the Industrial Area. (See Appendix B.) Figure 3-7 shows the estimated maximum fenceline impacts for U-235.

Finally, the predicted maximum fenceline concentration for U-238 was 1.91×10^{-8} pCi/m^3, and this was estimated to occur approximately 200 m to the south of the maximum for Am-241. Primary U-238 source areas in Site surface soils are located southwest of the Industrial Area, and are shown in Appendix B. Figure 3-8 shows the estimated maximum impacts for U-238.

3.4.2 Dose

The results of the concentration modeling were converted to effective dose equivalent in units of mrem using the conversion approach that was described in Section 3.3. Table 3-3 lists the maximum estimated dose for each actinide and the conversion factor that was applied to convert from pCi/m^3 to mrem.

Table 3-3. Conversion of Maximum Concentration to Dose

Isotope	Maximum Estimated Concentration (pCi/m^3)	Factor for Conversion of pCi/m^3 to mrem	Maximum Estimated Dose (mrem)
Pu-239	2.25×10^{-5}	5,000	1.13×10^{-1}
Am-241	3.64×10^{-6}	5,263	1.92×10^{-2}
U-233/234	7.82×10^{-9}	1,299	1.02×10^{-5}
U-235	1.87×10^{-9}	1,408	2.63×10^{-6}
U-238	1.91×10^{-8}	1,205	2.30×10^{-5}

Notes:
Am = americium
m = meter
pCi/m^3 = picocuries per cubic meter
Pu = plutonium
U = uranium

3.4.3 Deposition

Annual deposition rates of the five actinides were estimated with the ISCST3 model in units of pCi/m^2/yr. Figure 3-9 graphically presents the annual deposition of Pu-239 at the Site. As would be expected given the predominant winds at the Site, the annual deposition contours extend toward the east-southeast from the east edge of the Industrial Area and the 903 Pad Area. The contour representing 5.0 pCi/m^2/yr deposition rate extends approximately 1 km beyond the eastern fenceline of the Site. Centroids of Pu-239 surface soil concentrations are shown in Appendix B.

Figure 3-7. Model Predicted Annual Concentration of Activity for U-235

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Annual Concentration of Activity
(pCi/m^3) for U-238

- 7.00E-010 to 2.54E-009
- 2.54E-009 to 4.38E-009
- 4.38E-009 to 6.22E-009
- 6.22E-009 to 8.06E-009
- 8.06E-009 to 9.90E-009
- 9.90E-009 to 1.17E-008
- 1.17E-008 to 1.36E-008
- 1.36E-008 to 1.54E-008
- 1.54E-008 to 1.73E-008
- 1.73E-008 to 1.85E-008
- 1.85E-008 to 1.91E-008

0.5 0 0.5 1 1.5 Kilometers

NOTE:
Coordinates reference
UTM Meters, Zone 13, NAD27

Figure 3-8. Model Predicted Annual Concentration of Activity for U-238

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N

Annual Deposition of Pu-239 (pCi/m^2)
5
105
205
305
405
505
605
705
805
Installation Boundary

0.5 0 0.5 1 1.5 Kilometers

NOTE:
Coordinates reference
UTM Meters, Zone 13, NAD27

Figure 3-9. Model Predicted Annual Deposition of Pu-239

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Figure 3-10 presents the annual deposition rate of Am-241. The contour pattern is similar to that for Pu-239, but the overall magnitudes of the contours are lower. The 1.0 $pCi/m^2/yr$ deposition rate for Am-241 follows a similar contour to that for a 5.0 $pCi/m^2/yr$ deposition rate for Pu-239. Appendix B shows the centroid locations of surface soil concentrations of Am-241.

The patterns of annual deposition (relative to each other or to Pu-239 or Am-241 sources) for the uranium isotopes were variable because of the differing locations of the source areas (see Appendix B), coupled with prevailing westerly to northwesterly winds on an annual basis (see Figure 3-1). The overall magnitudes of the uranium deposition rates were much lower than the deposition rates for Pu-239 and Am-241, as would be expected given the relative magnitudes of activity in the soil. Figures 3-11 through 3-13 present the annual deposition rates for U-233/234, U-235, and U-238, respectively.

Figure 3-11 shows that the contour representing 0.01 $pCi/m^2/yr$ for U-233/234 is centered near the northeast part of the Industrial Area, and encloses an area that is approximately the size of the Industrial Area itself. As discussed previously, the major surface soil concentration areas for U-233/234 are near the Solar Ponds and may be due to contributions from naturally occurring uranium deposits. Figure 3-12 shows that a 0.01 $pCi/m^2/yr$ contour for U-235 is also located near the northeast part of the Industrial Area, but encloses a slightly smaller area than for U-233/234. Another area of at least 0.01 $pCi/m^2/yr$ deposition rate for U-235 is located just to the southwest of the Industrial Area. These contours reflect U-235 surface soil sources near the Solar Ponds and southwest of the Industrial Area.

For U-238, Figure 3-13 shows that a 0.01 $pCi/m^2/yr$ contour is also centered just to the southwest of the Industrial Area, but covers a larger area that extends nearly to the eastern fenceline of the Site. The primary surface soil concentrations of U-235 are generally southwest of the central portion of the Industrial Area.

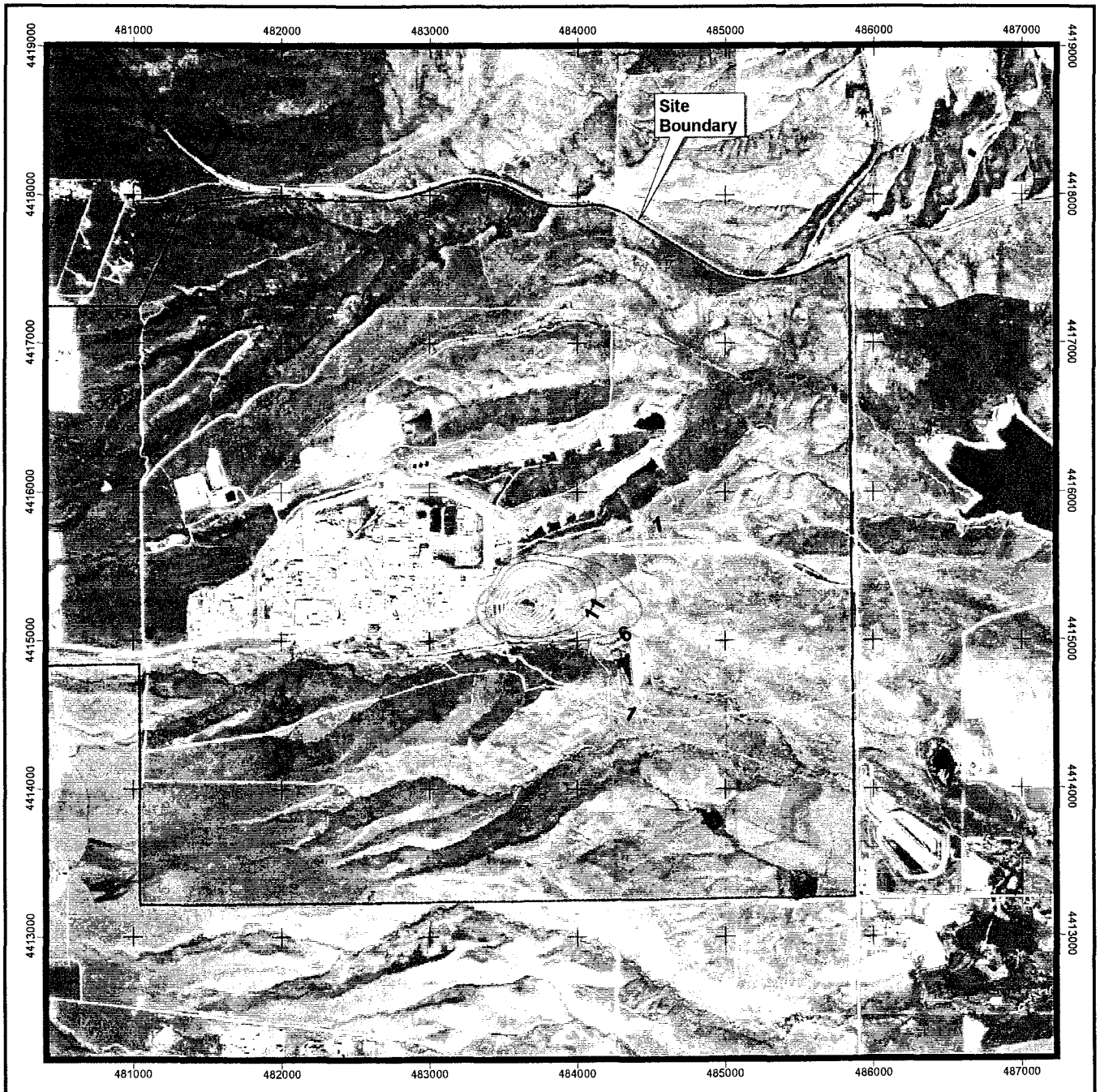

Figure 3-10. Model Predicted Annual Deposition of AM-241

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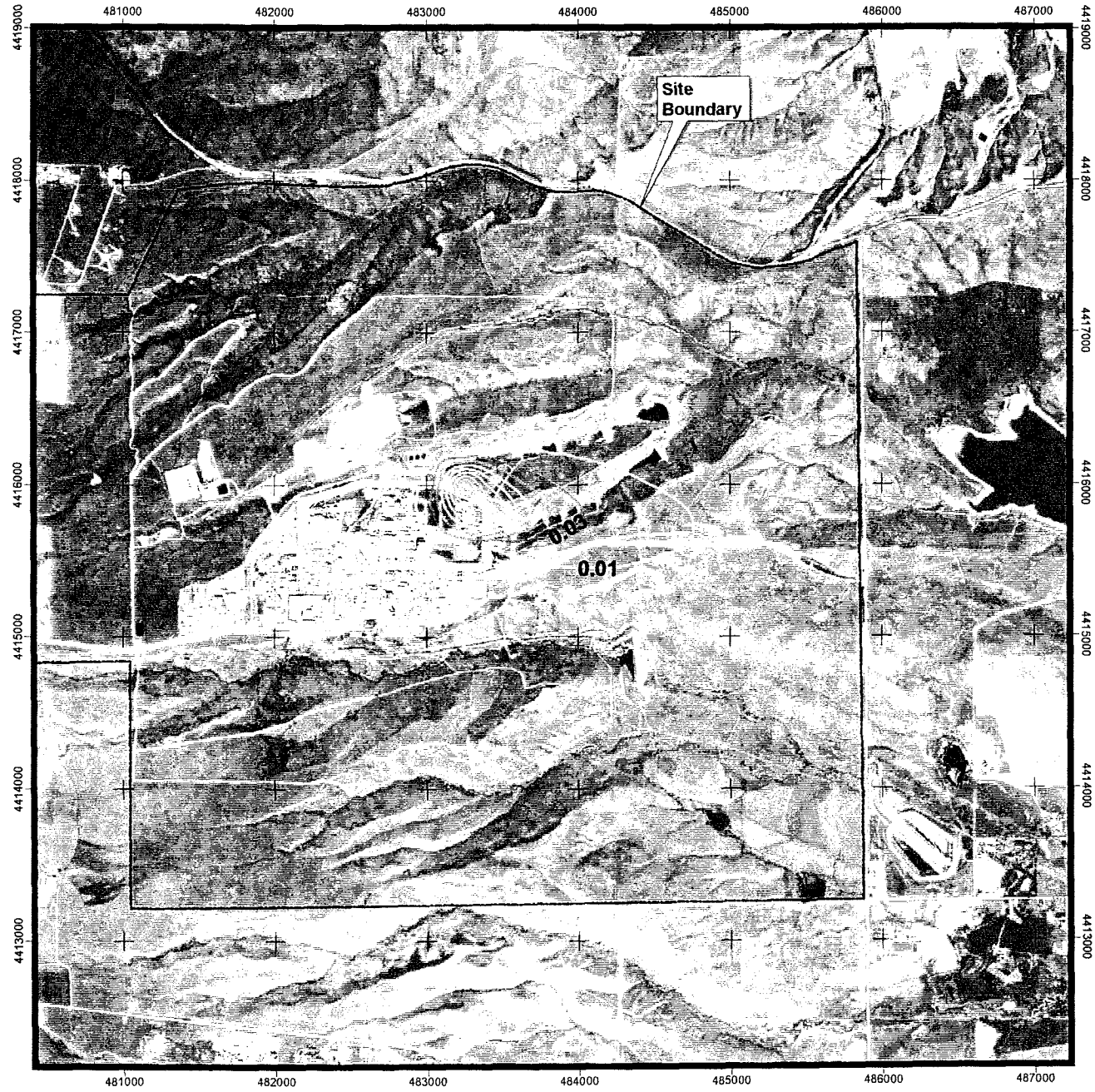

N

Annual Deposition of U-233/234 (pCi/m^2)
0.01
0.03
0.05
0.07
0.09
0.11
0.13
0.15
Installation Boundary

0.5 0 0.5 1 1.5 Kilometers

NOTE:
Coordinates reference
UTM Meters, Zone 13, NAD27

Figure 3-11. Model Predicted Annual Deposition of U-233/234

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N

Annual Deposition of U-235 (pCi/m^2)
0.01
0.03
0.05
0.07
0.09
0.11
0.13
Installation Boundary

0.5 0 0.5 1 1.5 Kilometers

NOTE:
Coordinates reference
UTM Meters, Zone 13, NAD27

Figure 3-12. Model Predicted Annual Deposition of U-235

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Figure 3-13. Model Predicted Annual Deposition of U-238

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4.0 COMPARISON AND SENSITIVITY ANALYSES

This section discusses comparison studies that were performed to "validate" the emission estimates and model results based on measured air quality data from the Site Radioactive Ambient Air Quality Monitoring Program (RAAMP) samplers. In addition, the uncertainty associated with the model and emission estimation methods was explored through several sensitivity analyses that varied factors and assumptions that affect emissions, deposition, or dispersion. The information reported in this section will be used to refine model input data for future work.

4.1 Comparison Studies

There are many potential sources of uncertainty in the analyses reported in Sections 2.0 and 3.0 of this report. As discussed in Section 2.2, the resuspension of actinides is a complex process that is variable over time and highly site specific. It also does not lend itself well to direct observation and measurement under ambient conditions. Of the many specific factors that could influence resuspension that were identified in the conceptual model, only the major factors (wind speed, snow cover, and the distribution of actinides in Site surface soils) were directly accounted for in the method used to predict emissions. Although development of the emission estimations relied heavily on Site-specific data that accounted for some of the other influences acting on resuspension (e.g., surface characterization), many sources of uncertainty remain.

Similarly, the modeling system represents a simplification of reality. Both the ISCST3 model itself and the specific inputs used to represent Site emissions rely on a variety of assumptions that may not entirely correspond to actual conditions.

A comparison of model results with actual measured concentrations of actinides at the Site was performed to provide data regarding the overall accuracy of the model predictions. The measured data provide some perspective on the relative accuracy of the model, but are insufficient to confirm specific factors contributing to discrepancies between predicted concentrations and measured concentrations.

4.1.1 Data Available for Comparison with Model Results

The Site operates an ambient sampler network that collects particulates on filter cartridges (the RAAMP network). Samplers are located at intervals along the Site fenceline, at a number of locations within the Buffer Zone and Industrial Area, and in surrounding communities. The cartridges are generally exchanged monthly and either archived for possible future analysis or analyzed immediately for specific radioactive isotopes, depending on the sampler location.

During 1996, samples from three locations were subjected to routine isotopic analysis. Two of the locations were along the southeast fenceline, generally downwind from the 903 Pad. The locations of the three fenceline samplers, designated S-038, S-138, and S-207, are shown in Figure 4-1. Sampler S-038 is an older style sampler that was

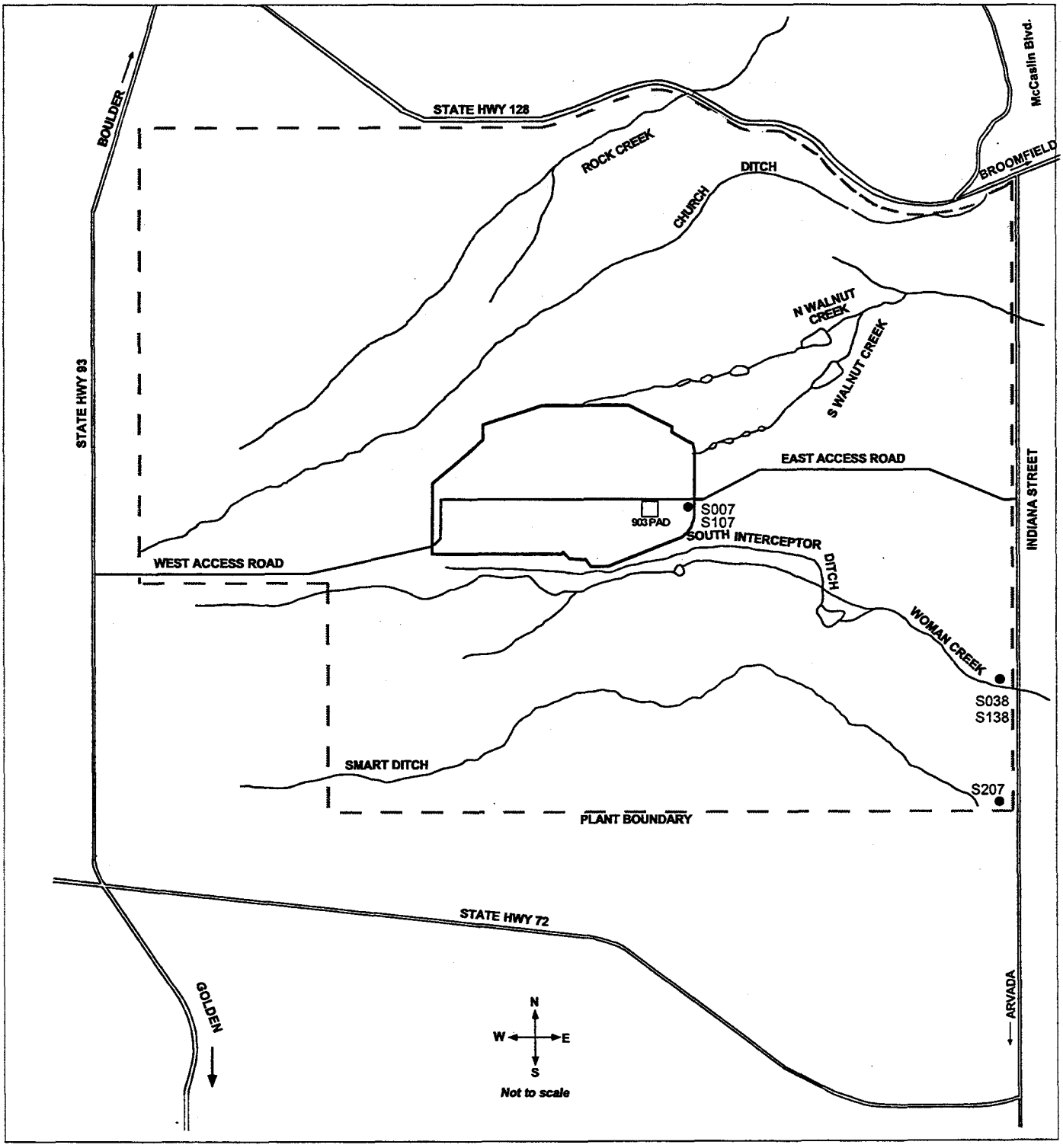

Figure 4-1. Location of Samplers Used for Comparison Studies

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collocated with S-138. S-038 cartridges were exchanged twice a month and the samples were composited (combined) and analyzed monthly for Pu-239. Cartridges from S-138 and S-207 were exchanged monthly. The samples were composited at each location and analyzed quarterly for Pu-239 until October 1996, when monthly analyses were begun.

In addition to the fenceline samplers, two collocated samplers were located within the Buffer Zone just downwind of the 903 Pad. These samplers, designated S-007 and S-107, are also shown in Figure 4-1. S-007 was an older style sampler that was exchanged and analyzed for Pu-239 on the same schedule used for S-038. Similarly, samples at S-107, a newer sampler, were exchanged and analyzed on the same schedule as S-138 and S-207.

To compare model results against measured data, the model described in Section 3.0 was run for the 1996 sampling periods. Concentrations were estimated at receptors along the fenceline and at the approximate location of samplers S-007/S-107. The comparison data are discussed in Section 4.1.2.

Because only natural resuspension emissions were modeled in this study, other sources of Pu-239 emissions from the Site during the same timeframe could skew the measured data such that the results would not be comparable. However, it is expected that other Pu-239 emission sources did *not* contribute significantly to measured 1996 concentration data, as explained below.

Site-wide radionuclide emissions are estimated annually for regulatory reporting purposes. In general, unless a project or activity has resulted in a nonroutine radionuclide release, Site radionuclide emissions have been dominated in recent years by natural resuspension from contaminated soil areas. During 1996, two projects, the Trench 3/Trench 4 (T-3/T-4) source removal project and the 774 clarifier tank draining project, produced small but nonroutine radionuclide emissions. The radionuclide emissions from T-3/T-4 were primarily U-238, however, and would not have affected Pu-239 measurements to any great extent.

The 774 tank draining did release small amounts of Pu-239; however, the release was a very short-term event of less than a week duration in early August 1996. The tank was located northwest of the samplers examined in this study and analysis of filters from samplers surrounding the tank showed elevated Pu-239 levels to the east but not to the south, west, or northwest. Therefore, releases from this event would have had little impact at S-007/S-107, S-038/S-138, or S-207 and, in fact, Pu-239 concentrations measured at those locations during August were among the lowest measured during 1996. As a result, it is thought that Pu-239 concentrations measured during 1996 were again dominated by natural resuspension from around the 903 Pad at the specific sampling locations used for this comparison.

4.1.2 Results of Modeling Comparison Studies

This section compares measured and modeled actinide concentrations at samplers located in the Buffer Zone and around the Site fenceline.

Buffer Zone Samplers

As discussed previously, concentrations of Pu-239 were predicted at the location of the S-007/S-107 samplers using the modeling system developed for the Actinide Migration Evaluation. Concentrations were estimated for the discrete periods during 1996 for which individual samples were collected.

To predict concentrations for the sampler location, an array of five receptors was used. One receptor was located at the known location of the samplers, and the other four receptors were located 25 m to the north, south, east, and west of the known location. An array of receptors was used to determine if predicted impacts within the vicinity of the monitor location would vary greatly. A review of the model results indicated that the predictions were consistent, and therefore the impact at S-007/S-107 was computed by averaging the model results for the five receptors in the array.

The measured and modeled concentrations are shown in Figure 4-2. Model predicted concentrations are higher, by one-to-two orders of magnitude, than the measured concentrations. The data collected by S-107 supports the temporal trend in predicted concentrations. Such trend is not supported by the S-007 data, however. The results of the modeling for the various sampling periods are presented in Table 4-1.

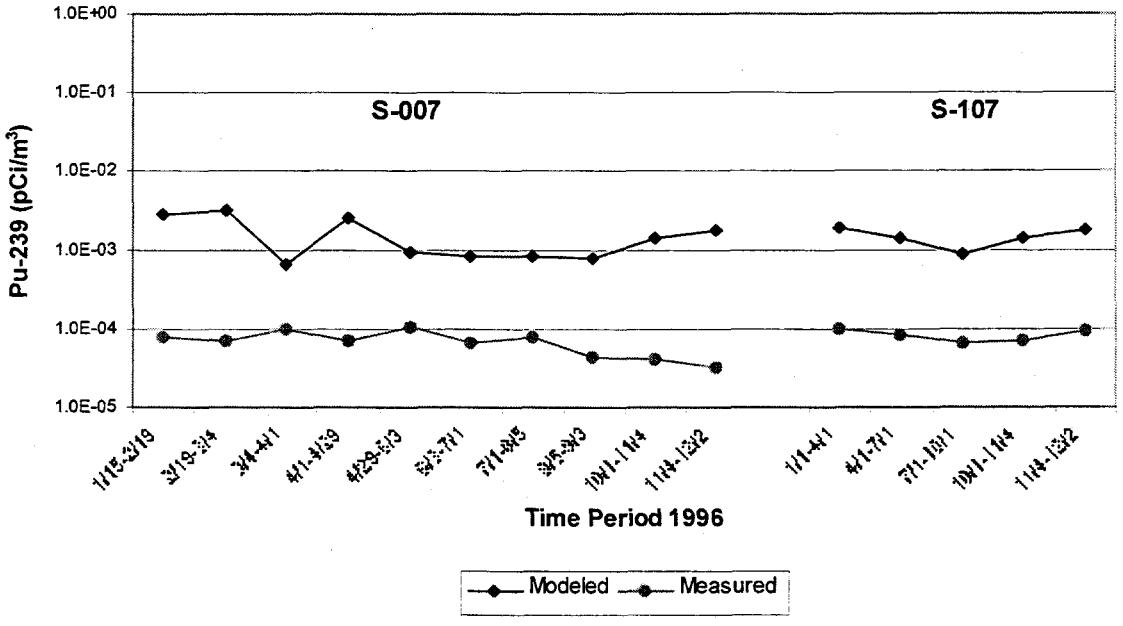

Figure 4-2. Comparison of Modeled and Measured Pu-239 Concentrations

Table 4-1. Results of ISCST3 Modeling at S-007/S-107 and Measured Levels of Pu-239

Sampling Period in 1996	Modeled Activity (pCi/m^3)	Measured Activity (pCi/m^3)
S-007		
1/15-2/19	2.74×10^{-3}	7.60×10^{-5}
2/19-3/4	3.24×10^{-3}	7.10×10^{-5}
3/4-4/1	6.51×10^{-4}	9.60×10^{-5}
4/1-4/29	2.49×10^{-3}	6.90×10^{-5}
4/29-6/3	9.21×10^{-4}	1.04×10^{-4}
6/3-7/1	8.17×10^{-4}	6.60×10^{-5}
7/1-8/5	8.31×10^{-4}	8.01×10^{-5}
8/5-9/3	7.61×10^{-4}	4.24×10^{-5}
10/1-11/4	1.40×10^{-3}	4.12×10^{-5}
11/4-12/2	1.76×10^{-3}	3.21×10^{-5}
S-107		
1/1-4/1	1.91×10^{-3}	1.01×10^{-4}
4/1-7/1	1.39×10^{-3}	8.11×10^{-5}
7/1-10/1	8.46×10^{-4}	6.44×10^{-5}
10/1-11/4	1.40×10^{-3}	7.11×10^{-5}
11/4-12/2	1.76×10^{-3}	9.50×10^{-5}
Annual Average at S-007	1.49×10^{-3}	6.83×10^{-5}
Annual Average at S-107	1.49×10^{-3}	8.18×10^{-5}

Notes:

pCi/m^3 = picocuries per cubic meter

The emission estimation method used (described in Section 2.3 of this report) assumed that the activity in soil on vegetation surfaces (the soil available for resuspension) was the same as the activity of the underlying soil. If predicted actinide concentrations are adjusted to account for the fact that the activity of soil on vegetation is likely "diluted" with respect to the activity of the underlying soil (see Section 2.2.1), then model predictions are within an order of magnitude of the measured actinide concentration data (e.g., reducing the predicted actinide concentrations to 20% of their original value yields impacts that are within an order of magnitude of the measured data).

Fenceline Samplers

Model-predicted levels of activity concentration for Pu-239 at the Site fenceline were compared to measured levels of activity from fenceline samplers. Annual average concentrations in pCi/m^3 were calculated for two samplers that are located along the east Site fenceline: 1) sampler S-138, which is located on a east-southeast vector from the

Industrial Area, and 2) sampler S-207, which is located on a southeast vector from the Industrial Area.

Table 4-2 presents the results of the comparison between the measured activity and the range of activity that was predicted at receptors within 100 m of the sampler locations. While the measured levels at both samplers are of the same magnitude, the modeled levels at S-138 are approximately one order of magnitude higher than those at S-207. As a result, the modeled concentrations at S-138 are approximately two orders of magnitude higher than the measured levels, while at S-207 the modeled levels were approximately one order of magnitude higher than the measured values. These differences in measured and modeled concentrations are similar to those seen with the Buffer Zone samplers. Again, if activity on vegetation is diluted relative to the underlying soil, the concentrations would be within approximately one order of magnitude.

Table 4-2. Results of ISCST3 Modeling at S-138 and S-207 and Measured Levels of Pu-239

Sampling Period	Modeled Activity (pCi/m^3)	Measured Activity (pCi/m^3)
S-138		
Annual Average for 1996	1.36×10^{-5} to 1.58×10^{-5}	1.94×10^{-7}
S-207		
Annual Average for 1996	2.33×10^{-6} to 4.58×10^{-6}	1.78×10^{-7}

Notes:
pCi/m^3 = picocuries per cubic meter

4.1.3 Discussion

The comparisons reported in Tables 4-1 and 4-2 show that modeled emissions from natural resuspension at the Site appear to overestimate measured concentrations by one-to-two orders of magnitude. Dilution of activity on vegetation surfaces, relative to activity in the underlying soil, may account for the better part of one order of magnitude difference. Other factors may include differences in current vegetation cover, and other factors, compared to conditions under which the wind tunnel studies that were used in estimating emissions were performed; inherent conservatism in the model formulation; and possible differences between what the RAAMP samplers measure and what was modeled. These factors are explored below.

Emission Estimation Factors

Most importantly, resuspension has been shown to be a highly variable and site-specific phenomenon. The emission estimation method detailed in Section 2.3 used the "best" data available to estimate emissions; however, the data set was quite small, which increases the uncertainty in the emission equation.

Vegetation cover in the areas surrounding the 903 Pad appears to be more dense (complete) than cover in the areas used in the wind tunnel study that forms the basis for the emission equation. Differences in vegetation cover could have the following effects:

- More complete vegetation cover would increase boundary layer resistance, which would increase the threshold wind velocity necessary to resuspend surface soil and litter. This may decrease overall resuspension rates and would also increase the relative importance of the reservoir of dust on vegetation surfaces as a source of resuspendable particulates. In addition, the size distribution of resuspended particles may be affected.

  A less vegetated surface, in contrast, would leave more bare soil area exposed; however, such areas may be "crusted over", which would serve to limit soil erosion. Further, a less vegetated surface would provide less potential for resuspension from the vegetation itself. As a result, it is uncertain whether the changes in vegetation that have occurred over time have increased or decreased resuspension rates.

- More vegetation would inhibit rainsplash from transferring large amounts of soil to plant surfaces. Instead, deposition of particulates from upwind areas would account for a larger percentage of the particulate loading on vegetation surfaces. In general, over the contaminated soil areas, rainsplash would transfer soils with higher actinide activity to vegetation than would deposition. As described previously, much of the particulate deposited would derive from clean upwind soil sources, thereby diluting activity on plant surfaces relative to the underlying soil.

Another factor to consider is particle size. The emission estimation approach assumed a constant particle size distribution. However, the size distribution of resuspended particles is actually dependent on wind speed: the higher the wind speed, the greater the fraction of larger particles resuspended. Data collected on Site indicate that coarse particles carry most of the actinide activity. As a result, the emission estimation approach using a constant particle size distribution could be overestimating the amount of larger particles resuspended, with a resulting overestimate in actinide concentrations.

Dispersion and Deposition Model Factors

As discussed in Section 2.1, Gaussian plume models, in general, offer a trade-off between precision and ease of use. The model inputs, including meteorological data, are readily available and the model set up can be easily modified to simulate different scenarios. However, the results may be considered to have an uncertainty that ranges from a factor of 2 to over an order of magnitude (EPA, 1995a).

Gaussian models have been studied and validated extensively and their behavior, the precision and accuracy of resulting predictions, is generally well understood. Longer-term averages (i.e., annual) have less uncertainty than short-term results (i.e., 1 hour) and patterns of concentration/deposition are more reliable than a single-point prediction. Models recommended by EPA for use in regulatory decision making (such as ISCST3)

are conservative; that is, under most circumstances, concentrations are more likely to be overestimated than underestimated.

ISCST3 is one of the most widely used of the regulatory air dispersion models developed and supported by EPA due to the model's ability to simulate dispersion from a variety of emission source types. The key features of ISC for this study were its area source and dry deposition algorithms. Both of these algorithms have undergone significant review and evaluation (EPA, 1989; EPA, 1992a; EPA, 1992b; EPA, 1992c; EPA, 1994).

In conclusion, ISCST3 has been independently validated. There is inherent uncertainty in Gaussian formulations and the type of sources and conditions dealt with in this study (areas sources, rolling terrain) increase uncertainty. ISC is known to be conservative and will more often overestimate than underestimate, which would contribute to the apparent overestimation of measured actinide values.

Other Factors

Another factor that could contribute to the apparent overprediction by the model would be any tendency by the RAAMP samplers to underrepresent the mass of particulate in the air. No ambient sampler is 100% efficient; some undercollection would be expected. Based on limited testing by EPA of the newer sampler design in use at the Site, it is expected that this undercollection is relatively small, less than 20 percent. While small, undercollection by the sampler would represent another source of discrepancy between modeled and measured actinide concentrations.

Samplers also differ in the size fraction collected. The RAAMP samplers in current use at the Site are designed to collect both a fine fraction (generally up to approximately 10 μm diameter) and a coarse fraction (above 10 μm diameter), with an upper size cut off below 30 μm. In contrast, the particle size range resuspended in the wind tunnel studies from which the emission estimation equation was derived may have included a significant fraction above 30 μm (EG&G, 1994). This difference in the functional size ranges sampled by the various devices would also lead to overprediction of RAAMP data by the emission equation based on wind tunnel data.

Finally, the older style samplers (such as S-007 and S-038) are thought to have collected less particulate at high wind speeds than the newer sampler design. Based on several years of data from collocated monitors, the difference in collection efficiency only becomes significant when the monthly average wind speed exceeds 11 mph (see Appendix C to this report). This difference is therefore unlikely to be significant overall but it indicates that sampling data from S-007 and S-038 may have underrepresented peak actinide emission periods relative to the modeling analysis.

4.2 Sensitivity Analyses

A number of sensitivity analyses were performed to explore the effect that different assumptions might have on model results. These analyses are described below.

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4.2.1 Pu-239 Concentrations Using Background (Fallout) Activity Level

Isopleth maps showing surface soils concentrations of various actinides at the Site formed the basis for estimating emissions due to chronic, natural resuspension, as explained in Section 2.3. While the original intention in this study was to model resuspension from any areas of the Site where actinides were present above background (defined as regional fallout levels), the lowest isopleth cutoff for each actinide was somewhat above the fallout level. To determine whether the omission of larger areas of lower concentrations of actinides would have a significant effect on model predictions, another model run was designed, incorporating an additional area source reflecting the background, or "fallout" level of Pu (Pu was modeled because it has the largest contribution to total dose of the five isotopes considered in this study).

For this model run, a single area source defined by the Site boundary was established. An hourly emission file for this source was created that accounts for wind speed-dependent soil resuspension in the same manner as used for the primary Pu-239 model runs, but with an activity level for Pu-239 of 0.066 pCi/g (Chromec, 1999).

The maximum modeled Pu activity concentration due to this source alone was 4.3×10^{-7} pCi/m^3, which is approximately two orders of magnitude lower than the maximum concentration predicted for the primary Pu concentration run, reflecting the multiple, above-background, soil activity levels modeled. The predicted concentrations from this model run may be considered as estimates of potential exposure to wind-resuspended Pu in the absence of the Site.

Figure 4-3 shows the distribution of predicted activity for this model run.

4.2.2 Pu-239 Concentrations Using Langer's Constant Resuspension Rate

The model inputs used to estimate the annual concentration of Pu-239 were also used to estimate the concentrations that would result from emissions derived from a constant soil resuspension rate developed by Gerhard Langer. Past research on Site led to the development of a Site-specific resuspension rate of 2×10^{-12} sec^{-1} for the entire 903 Field area (Langer, 1991). As discussed in Section 2.2 of this report, this rate has been used in developing actinide emission estimates in support of the Site's annual reporting on radiation dose to the public required under 40 CFR 61, Subpart H.

Figure 4-3. Pu-239 Concentrations Using Background (Fallout) Activity Level

ACTINIDE MIGRATION STUDY
ROCKY FLATS ENVIRONMENTAL TECHNOLOGY SITE
GOLDEN, COLORADO

APPLICATION	DATE	DR. BY
ArcView GIS	09/10/99	JLC

To obtain the area source emission fluxes needed for modeling, the constant resuspension rate was applied in the following manner:

$$EF = (A)(2.5)(0.2)(10.000)\left(\frac{2x10^{-12}}{\text{sec}}\right)$$

where:

EF is the emission flux in units of $pCi/m^2/sec$;
A is the soil activity level in units of pCi/g;
2.5 is the particle density in units of g/cm^3; and
0.2 is the assumed depth of soil erosion.

Based on this emission estimation approach, the maximum modeled Pu activity concentration at the fenceline was 6.0×10^{-7} pCi/m^3, which is approximately two orders of magnitude lower than the maximum level predicted for the primary concentration run for Pu-239 (based on wind speed-dependent emissions) and similar in magnitude to measured activity levels at fenceline samplers. Figure 4-4 shows the distribution of activity concentration for this model run. Predictions were limited to property boundary receptors.

4.2.3 Comparison of Particulate Concentrations With and Without Plume Depletion

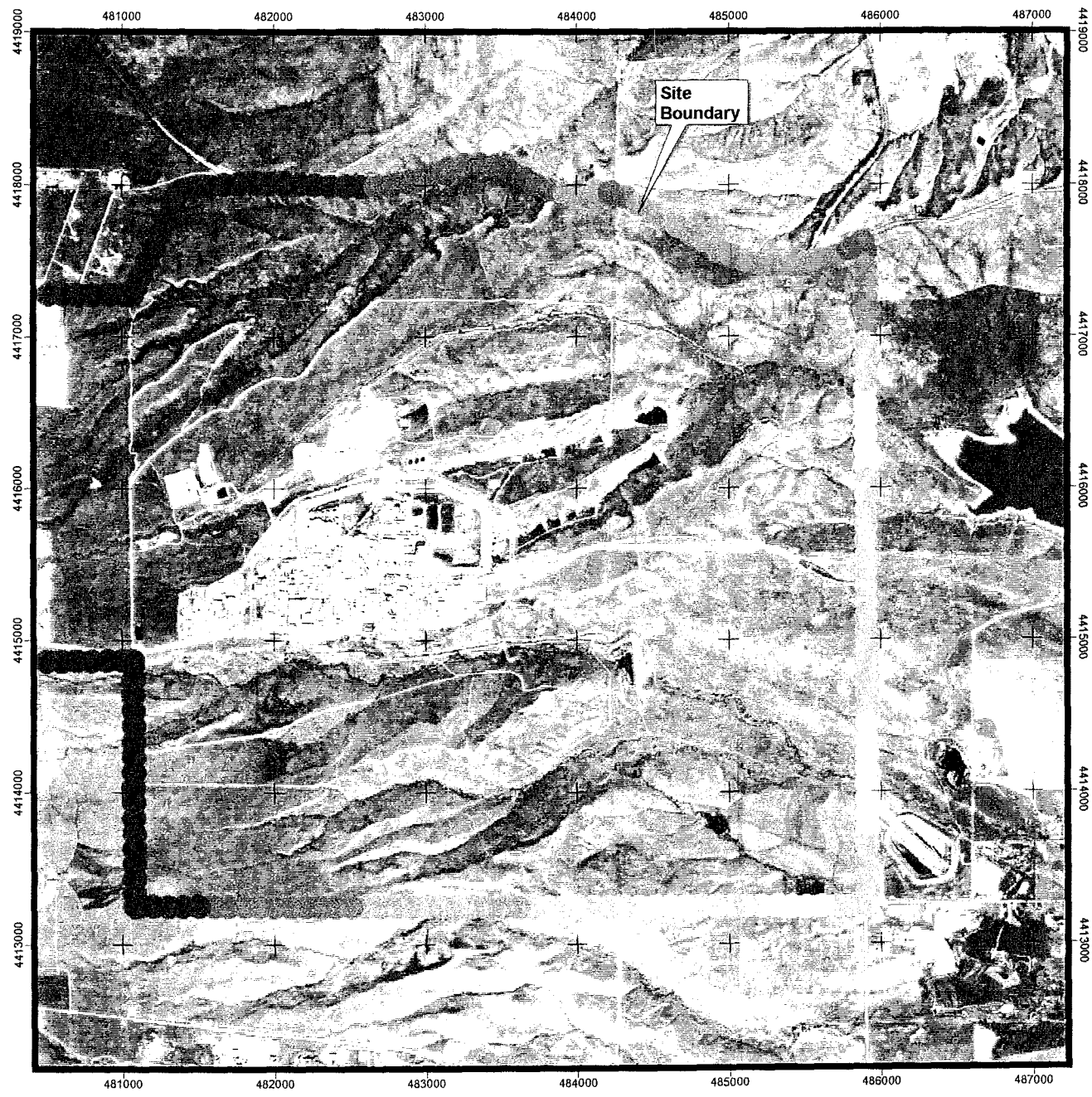
The final model sensitivity runs were developed to compare:

1) The ambient concentrations of particulate that would be predicted at the fenceline considering removal of pollutant mass from the plume due to dry deposition mechanisms, and

2) The levels that would result without considering such removal.

The primary model runs did not deplete the plume due to deposition of mass. As a result, the amount of activity predicted at the Site boundary would have been overestimated. These sensitivity runs were designed to provide a rough estimate of the amount of such overestimation.

Plume depletion was not accounted for with the primary model runs because the emissions for the primary runs were in units of activity and not mass. The ISCST3 model does not use information about particle sizes or density in dispersion calculations, only in deposition. As a result, resuspended activity could be modeled directly to predict activity concentration/dose, which greatly simplified (and speeded up) the analyses. Deposition calculations, in contrast, required separate runs for each particle size class.



Annual Concentration of Activity
(pCi/m ^{3}) for Pu-239 [Langer constant rate]

- 3.00E-008 to 8.20E-008
- 8.20E-008 to 1.34E-007
- 1.34E-007 to 1.85E-007
- 1.85E-007 to 2.37E-007
- 2.37E-007 to 2.89E-007
- 2.89E-007 to 3.41E-007
- 3.41E-007 to 3.93E-007
- 3.93E-007 to 4.45E-007
- 4.45E-007 to 4.96E-007
- 4.96E-007 to 5.84E-007
- 5.84E-007 to 6.00E-007

N

0.5 0 0.5 1 1.5 Kilometers

NOTE:
Coordinates reference
UTM Meters, Zone 13, NAD27

Figure 4-4. Pu-239 Concentrations Using Langer's Constant Resuspension Rate

ACTINIDE MIGRATION STUDY
ROCKY FLATS ENVIRONMENTAL TECHNOLOGY SITE
GOLDEN, COLORADO

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Plume depletion serves to remove mass from the plume, thereby decreasing predicted ambient (mass) concentrations. For this comparison, a subset of the Site boundary receptors was chosen along the east fenceline where maximum predicted annual impacts would be expected. An hourly emission file was created for resuspension of soil without regard to activity (i.e., emissions of particulate mass only), and annual concentrations were predicted at each receptor. The particle size parameters that were used for plume depletion were as follows:

- Category 1: 3 μm diameter (for <3 μm particles), mass fraction = 0.17, particle density = 2.5 g/cm^3;
- Category 2: 10 μm diameter (for 3-15 μm particles), mass fraction = 0.23, particle density = 2.5 g/cm^3; and
- Category 3: 15 μm diameter (for >15 μm particles), mass fraction = 0.60, particle density = 2.5 g/cm^3.

A single area source was used for the model runs. The shape of the area source matched that of the largest Pu-239 activity isopleth from the primary model runs. A total of 26 receptors was used to predict concentrations with and without plume depletion. At each receptor, the predicted concentration in μg/m^3 was lower with plume depletion by at least 20%, and at most 26%. The average decrease in predicted concentration with plume depletion for all 26 receptors was 24%.

Although source configurations between the primary and sensitivity runs differed, the analysis shows that the activity-based primary runs may have significantly over-predicted activity concentrations. This would partially account for the differences seen between measured activity levels and those predicted with the modeling analysis. For future model runs, plume depletion should be considered in calculating ambient concentrations.

4.2.4 Discussion

The sensitivity analyses show that the inclusion of additional isopleth data at lower levels would not significantly alter maximum Pu-239 concentration/dose predictions. It may be assumed the same would be true of the other actinides because fallout levels for Am-241 and uranium isotopes represent a smaller percentage of the lowest isopleth compared to Pu-239.

Use of Langer's constant resuspension rate would produce lower activity concentration predictions than the emission estimation method used in this study. As a result, concentration predictions based on Langer's method more closely match measured data. These results make sense in light of the fact that the constant resuspension rate was based on ambient wind data collected from a portable station previously located near the 903 Pad, and particulate data collected from the vertical dust flux tower, also previously located east of the 903 Pad (Langer, 1986).

The sensitivity analysis that examined plume depletion suggests that this may be a factor in model overpredictions. This should be taken into account in future modeling analyses.

4.3 Related Information

This section discusses three additional investigations that are related to the comparison and sensitivity studies discussed above.

4.3.1 Sampling and Meteorological Data Correlation

As part of an independent effort conducted by the Site air monitoring program, long-term measured actinide concentration data have been compared with meteorological data to determine if they are correlated. The work was prompted by apparent increases in measured concentrations of Pu-239 at the S-107 sampler (relative to annual average concentrations) during January 1997 and 1998 that were not also observed at the collocated S-007 sampler. The increases could not be attributed to any known Site activities, such as soil or groundwater sampling, that may have disturbed soils in the vicinity of the 903 Pad. Examination of historical data from the S-107 and S-007 samplers showed increases in Pu-239 concentrations at those locations during winter or spring months in other years, as well.

The newer sampler design (S-107) is expected to collect and retain particulates more efficiently than the older style sampler during high wind events. As discussed in Section 2.0, the soils at the Site, if undisturbed, have a high threshold wind speed for resuspension (although low, chronic levels of resuspension are thought to occur from vegetation surfaces at much lower wind speeds). Consequently, high wind events that may resuspend surface soil and litter, as well as dust from vegetation surfaces, are thought to be responsible for a significant portion of the natural actinide resuspension that occurs on Site. High winds occur at the Site during frontal passages and particularly during chinook conditions, when the warm, dry downslope winds can result in sustained winds above 35 mph and gusts over 100 mph. These conditions typically occur sporadically from January through April and, less commonly, during fall months.

Monthly Pu-239 concentration data from S-007 and S-107 were plotted against average and peak monthly wind speed data (as measured at the on-Site meteorological tower); westerly "fetch" (calculated as the average monthly wind speed times the percent of winds from the west, west southwest, or west northwest); and westerly fetch divided by total monthly precipitation. Apparent correlations in the data were further examined using several statistical tests. A more complete description of this analysis is included as Appendix C to this report.

The statistical tests showed:

- There is a statistically significant difference (at the 95% confidence level) between Pu-239 concentrations when westerly fetch exceeded 400 and when westerly fetch was below 400.

- In contrast, there is no statistically significant difference in Pu-239 concentrations when westerly wind fetch divided by precipitation exceeded 1300 compared to values below 1300. The lack of difference is true for all data and for just S-107 data; the test statistic was very close to zero for all combinations evaluated.

- When S-007 and S-107 were running simultaneously, the Pu-239 concentrations at S-107 became significantly different from the concentrations at S-007 only when the average monthly wind speed exceeded 11 mph. This appears to be the average threshold at which the new sampler design becomes more efficient than the older design at capturing airborne particulates.

The comparison shows that strong, westerly winds are correlated with higher Pu-239 concentrations to the east of the 903 Pad. In contrast, precipitation does not affect this relationship. Based on emission investigations performed by Gerhard Langer at the Site during the 1980's (discussed in Section 2.0), actinides are resuspended from Site vegetation surfaces even when the ground is saturated; only snow cover causes resuspension to cease. The lack of correlation with precipitation data *per se* supports this observed effect.

The data comparison also confirms that the newer sampler design is more efficient at collecting particles than the older design during high wind events. However, previous statistical comparisons using all available data from the S-007/S-107 and S-138/S-038 paired samplers has shown that there is no statistically significant difference in Pu-239 concentrations between old and new style samplers when data at all winds speeds are compared.

4.3.2 Representativeness of Meteorological Data

One important question concerning the accuracy of model predictions over the long term is whether or not 1996 was a "typical" weather year. Because both emission and dispersion calculations depend heavily on wind speed, this parameter is of particular interest. Snow cover was also taken into account in determining resuspension emissions, and wind direction (along with other meteorological factors such as temperature, mixing height, and stability) is used in the dispersion and deposition algorithms.

The Site has collected meteorological data for many years. Long-term historical average values for precipitation, wind speed, and wind frequency distribution have been compiled previously (Aerovironment, 1995) and were used for comparison with 1996 data.

Figure 4-5 compares 1996 monthly precipitation with monthly mean and median precipitation for the period 1953 through 1977 and 1987 through 1993. As described in Section 4.3.1, precipitation amounts do not exhibit a strong correlation with measured actinide concentrations near the 903 Pad. However, snow cover has been shown to effectively eliminate resuspension. Consequently, the comparison of precipitation data

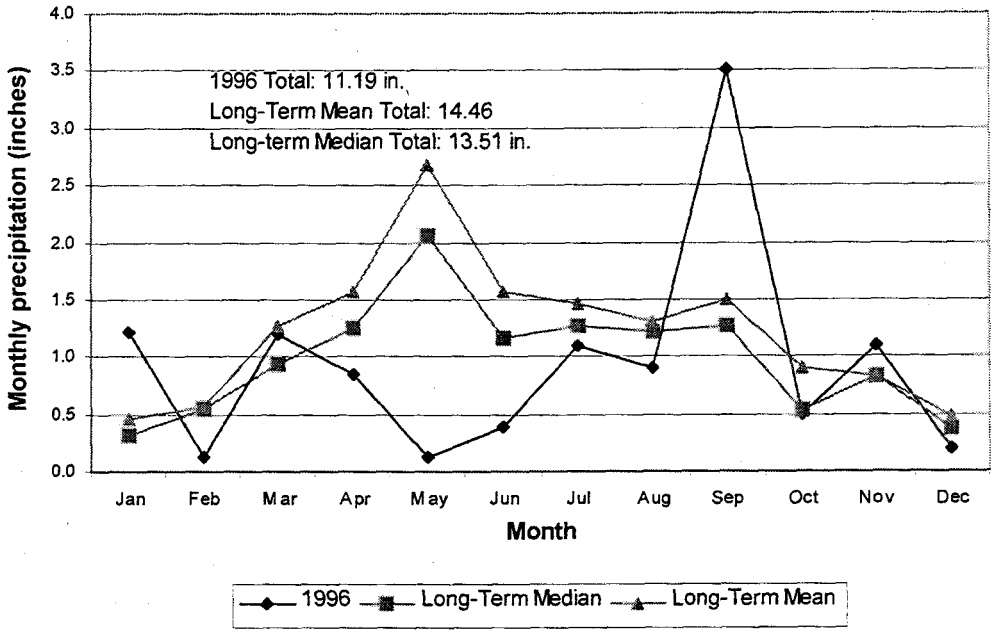

Figure 4-5. Comparison of 1996 Precipitation to Long-Term Site Data

shown in Figure 4-5 is most important for the winter, spring and (occasionally) fall months, when most precipitation occurs as snow.

Figure 4-5 shows that 1996 was a somewhat drier year than normal, although probably well within normal variation (semi-arid climates such as occur along the Front Range of Colorado experience large annual variations in precipitation). February, April, and May were below average precipitation months, while January, March, and especially September showed higher monthly totals than seen in the historical record. Albedo (surface reflectivity) data used in estimating hourly emissions, as described in Section 2.3.4, indicate that precipitation occurred as snow in January, February, March, April, September, November, and December 1996.

Figures 4-6 and 4-7 show a comparison of 1996 average and peak wind speeds with historical data from 1964 through 1977 and 1984 through 1993 for average values, 1953 through 1977 and 1984 through 1993 for peak values. Peak wind speeds during 1996 were consistently above long-term averages for all months. Average wind speeds in 1996 generally tracked historical trends, with some monthly variability apparent during winter and spring.

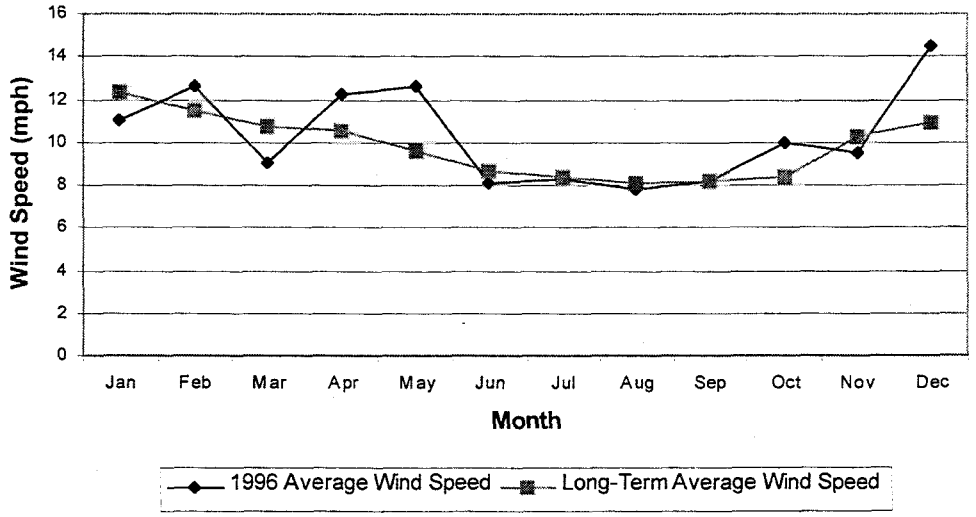

Figure 4-6. Comparison of 1996 Average Wind Speeds to Long Term Site Data

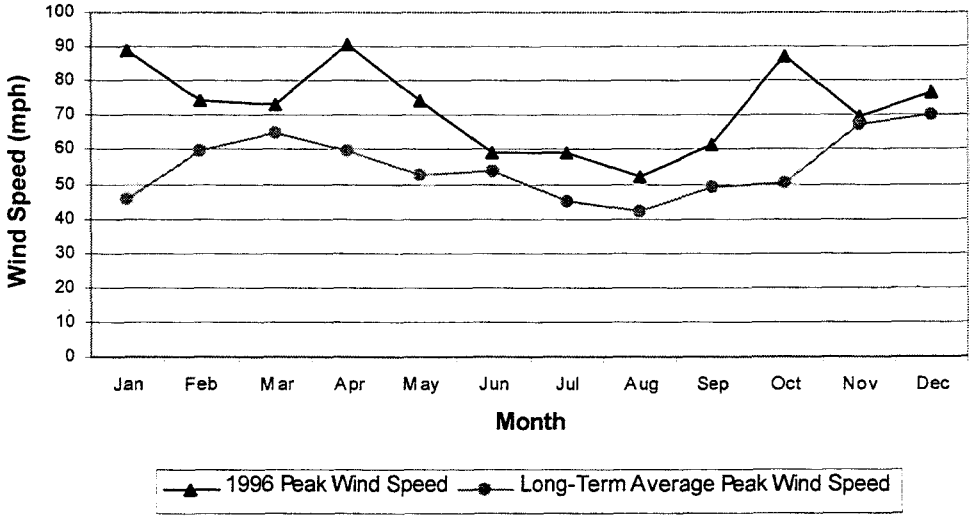

Figure 4-7. Comparison of 1996 Peak Wind Speeds to Long Term Data

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When the wind speed data are looked at in conjunction with the precipitation data shown in Figure 4-5, it appears that 1996 saw dry and windy conditions, alternating with periods of lower wind speeds and increased precipitation. Overall, 1996 may have resulted in somewhat enhanced resuspension relative to long-term averages, due to the conjunction of dry periods with higher average wind speeds, and the greater than average peak wind speeds over the entire year.

Figures 4-8 and 4-9 show a final comparison of the overall wind frequency distribution for 1996 and the long-term wind frequency distribution. Note that the historical graph shows ring values of 5, 10, 15, and 20% frequency, while the 1996 graph has corresponding ring values of 3, 6, 9, and 12 % frequency. On an annual basis, the wind frequency distribution for 1996 is quite consistent with the historical data.

4.3.3 Comparison of Deposition Algorithm Performance

Because ISCST3 performs numerical integration in its area source and deposition algorithms, computation times can sometimes become extreme. This occurred on this project due to the inclusion of the area source and dry deposition computations within the same runs (which is necessary for the simulation of actinide deposition from surface soil contamination). The latest version of ISCST3 (Version 99155), released very late in this study, has incorporated a non-regulatory TOXICS option that significantly reduces computation time under these circumstances. The reduction in computation time is accomplished by incorporation of a 2-point Gaussian quadrature routine for numerical integration for some situations, instead of the Romberg numerical integration used in the regulatory default mode. In addition, for area sources with dry depletion, another optimization routine is available to reduce model runtime by applying a single "effective" depletion factor to the undepleted area source integral, rather than applying the numerical integration for depletion within the area source integral.

A sensitivity analysis of the ISCST Version 99155 "short-cut" resulted in deposition predictons approximately 20% higher than the results using the previous version of ISCST3 (Version 98356). However, the new model completed in over 95% less time. This change provides the potential to perform more evaluative analyses in the future than could be performed for this study due to limitations imposed by the extreme computation times encountered.

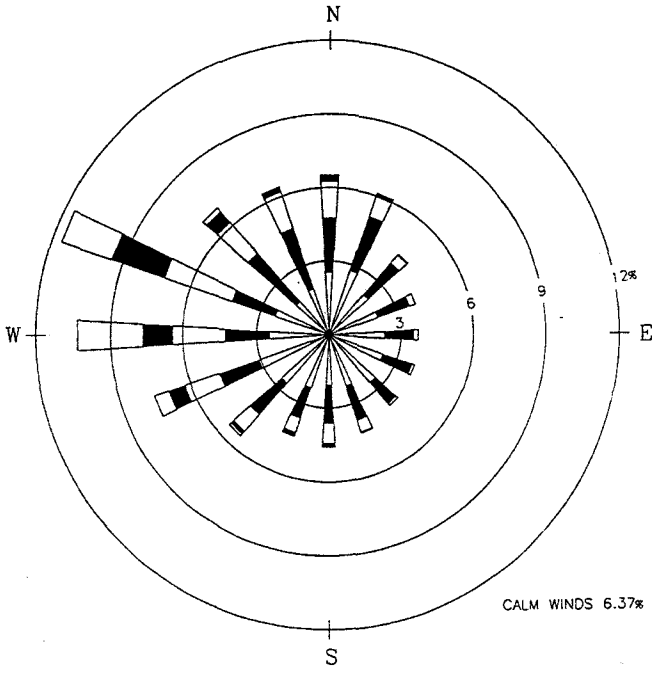

Figure 4-8. 1996 Wind Frequency Distribution

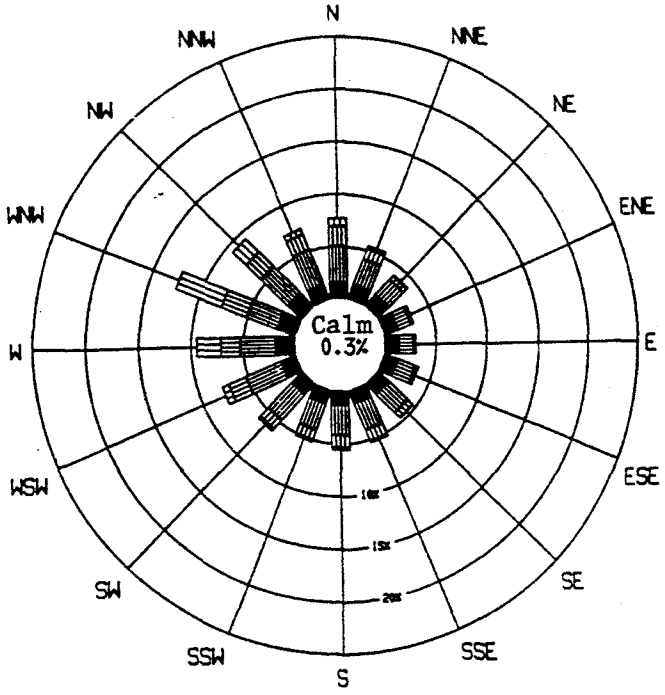

Figure 4-9. 1984-1993 Wind Frequency Distribution

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5.0 SUMMARY AND RECOMMENDATIONS

This section summarizes the air pathway work performed in FY99 and presents recommendations for future use of the work products developed.

5.1 Emission Estimation

Actinide resuspension due to natural phenomena at the Site is episodic in nature and influenced primarily by meteorological variables (wind speed and rainfall); particle and soil properties (moisture level and particle density); and surface characteristics (density and type of vegetative growth, and snow cover). Given the density of vegetation within the contaminated soil areas on Site, the primary source of contaminated soil resuspension is considered to be the dust-laden vegetation and litter, with little potential for direct resuspension from exposed soil surfaces (assuming no disturbance of the area).

A significant amount of research in particle and actinide resuspension has occurred over the years. This research emphasizes the need to customize any approach to the particular location of interest. The unique meteorological, soil, and surface characteristics for a given area must be taken into account to produce a reliable emission estimation approach. Past research conducted both at the Site and elsewhere indicate that soil resuspension can occur through a variety of mechanisms. The ability to express all known mechanisms quantitatively is unfortunately still limited due to insufficient data.

Past wind tunnel experiments on Site relate dust resuspension to ambient wind speed and currently provide the best method for estimating emissions. Site wind tunnel data indicate dust resuspension varies with wind speed raised to the third power.

An equation was derived relating hourly particulate and actinide emissions to wind speed, underlying surface-soil contamination levels, and the presence or absence of snow cover. This equation was used to calculate hourly emissions for five actinides due to natural resuspension mechanisms for the 1996 calendar year. The calculated emissions were used as input to dispersion and deposition simulations.

In addition, calculation methods were identified for a variety of anthropogenic emission mechanisms, such as excavation, traffic, maintenance of storage piles, etc. These methods will be used in conjunction with the natural resuspension equation identified above to calculate actinide emissions from specific remediation or D&D scenarios in future work.

5.2 Dispersion and Deposition Modeling

The ISCST3 model was configured to simulate dispersion from surface soil contamination areas at the Site. The hourly natural resuspension emissions calculated in the emission estimation task were input and the model was run using 1996 meteorological data. (1996 data were used because that year had more complete data capture at the on-Site towers for several parameters of interest, when compared with 1997 or 1998.) Air actinide concentrations, air dose, and deposition to ground or water surfaces were calculated at receptor locations along the Site fenceline and at 200-m intervals on and around the Site.

Maximum actinide concentrations were predicted to occur along the Site's eastern fenceline. For annual predictions this location was anticipated, given the predominant westerly winds at the Site. Similarly, the annual predicted Pu-239 and Am-241 deposition contours were found to extend toward the east-southeast from the eastern edge of the Industrial Area. The patterns of annual deposition for the uranium isotopes were variable because of the differing locations of the sources.

5.3 Comparison and Sensitivity Analyses

Air sampling data for Pu-239 were available for comparison with model results. Model predicted concentrations were found to be higher, by one-to-two orders of magnitude, than the measured concentrations (at both Buffer Zone and fenceline samplers). It is expected that the overprediction is partially due to dilution of the resuspendable dust attached to vegetation surfaces, relative to the actinide density in the underlying surface soils. Other potential factors were also identified.

Three sensitivity analyses were also performed to examine: 1) the inclusion of an additional source at the background or fallout level of Pu-239, 2) the performance of a general resuspension factor previously developed for the Site, and 3) the effect of plume depletion on predicted concentrations. The first analysis showed that inclusion of an additional source at background levels would not substantially increase predicted actinide concentrations. The second analysis showed that the Site-wide resuspension factor developed previously produces results (using the ISCST3 model) that match measured actinide concentrations fairly well (within the same order of magnitude). This is not surprising because the resuspension factor was developed from on-Site sampling data collected just to the east of the 903 Pad. However, the general resuspension factor can only be used to calculate annual average actinide values, whereas the method developed in this study can be used to vary emissions and impacts on an hourly basis.

The third analysis showed that removing the mass of particulate that is deposited to ground or surface waters from the plume would decrease predicted air concentrations (and dose) by 20 to 26 percent. The deposited particulate fraction, which was "double counted" in the study reported here, should be taken into account in future modeling.

Finally, information was presented on the representativeness of the 1996 meteorological data and on a related study that examined the strength of the correlation between meteorological variables and measured actinide air concentrations on Site. The 1996 year was fairly typical, although somewhat drier and windier than average, which indicates that actinide emissions due to natural resuspension were also probably somewhat above average in that year. However, windy conditions also enhance dispersion such that 1996 measured concentrations may or may not have exceeded average values.

Measured Pu-239 concentrations to the east of the 903 Pad were shown to be strongly correlated to the occurrence of strong, westerly winds (as expected). The amount of precipitation, on the other hand, did not directly correlate with measured concentrations.

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5.3 Recommendations for Future Modeling

The following recommendations apply to future modeling using the emission estimation method and dispersion/deposition model described in this report.

- Modeling of natural resuspension in future scenarios should consider possible activity dilution relative to the underlying soil. Field verification of this phenomenon may be useful.

- The sensitivity analysis that examined plume depletion suggests that this may be an important factor in the model overpredicting actinide concentrations. Therefore, a method to account for plume depletion in activity concentration/dose predictions should be developed for future modeling.

- Results of the sensitivity modeling appear to indicate that actinide surface soil concentrations below the existing isopleth levels may continue to be ignored in future modeling (their relative contribution is insignificant).

- Finally, the new TOXICS option is ISCST3 should be considered for future "what if" scenarios at least and performance relative to the existing algorithm should be examined further.

6.0 Bibliography

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Appendix A

Input Parameters for Fire Scenarios

Fires will be modeled in future scenarios as one or more area sources in ISCST3, with a release height above the ground of 0.0 m, an initial vertical dimension based on estimated plume rise (discussed below), and lateral dimensions based on the size of the burned area. The size of the burned area and other factors affecting plume rise would be preset for a prescribed burn but would be more variable for a wildfire scenario. The simulation may include one source area or multiple areas, depending on the overall size of the fire, the burn rate, and the location of the receptors of interest. Actinide and particulate emissions from each area will be based on unit emission estimations calculated as described in Sections 2.4 and 2.5.3.

The initial vertical dimension of the area source(s) will be estimated from plume rise data developed using a method described by Sestak and Riebau (1988) in the user's guide to the Simple Approach Smoke Estimation Model (SASEM). The method assumes that the heat produced by a fire line does not produce a single coherent plume. The average depth of the fire line is used as the characteristic dimension that determines what proportion of the heat of the fire acts to raise the plume along any part of its length. The length of the fire line is divided by the fire depth to obtain a number of "plumes" by which the line can be represented. The total heat output of the fire is then divided by the number of plumes to produce the heat output used for plume rise calculations.

The depth of the fire line is calculated as:

$$D = (FR)(RT)$$

where:

D is the depth of the fire line;
FR is the rate of spread of the fire; and
RT is the residence time of the fire.

Default residence times for various fuels are given in Sestak and Riebau (1988): 120 seconds for grass, 960 seconds for wood, and 480 seconds for sagebrush.

The rate of spread of the fire will be calculated from the assumed area of the burn and the fire duration (i.e., FR = length of side of burn/duration of burn). For a prescribed burn, these parameters would be a function of the burn plan. For a wildfire scenario, reasonable values will be determined in consultation with the Colorado State Forest Service or using fire behavior models such as BEHAVE (Andrews and Bevins, 1998).

Once the depth of the fire line is determined, the length of the fire line (a function of the size and shape of the area burned and the presumed direction of spread) is divided by the depth to calculate the number of plumes in the fire line (= NP).

The number of plumes is used to calculated the heat release for an individual plume:

$$QH = (HC)\left(\frac{F}{NP}\right)$$

where:

QH is the heat release for an individual plume (in calories per second, cal/s);
HC is the heat content of fuel (heat per mass burned);
F is the fuel consumption rate (mass fuel per unit area); and
NP is the number of plumes.

Default values for HC are also given in Sestak and Riebau (1988) for various fuels: 3.33 megacalories per kilogram (Mcal/kg) for grass, 3.88 Mcal/kg for wood, and 3.50 Mcal/kg for sagebrush.

Fuel consumption rate is calculated as follows:

$$F = (CF)(FL)\left(\frac{A}{T}\right)$$

where:

F is the fuel consumption rate;
CF is the fuel consumption factor (see Section 2.4; varies with fuel type);
FL is the fuel loading (mass per unit area; see Section 2.4);
A is the area burned; and
T is the fire duration.

A and T would again be a function of the burn plan or would be set as part of a wildfire scenario.

Once the heat release of each plume is determined, plume rise can be calculated. Sestak and Riebau (1988) give the following formula:

$$H = 0.0101\left(\frac{QH^{0.75}}{U}\right)$$ for stability A to D and QH < 1.4×10^6 cal/s

$$H = 0.0847\left(\frac{QH^{0.6}}{U}\right)$$ for stability A to D and QH > 1.4×10^6 cal/s

$$H = 0.917\ QH^{0.33}\ U^{0.33}$$ for stability E and F

where:

H is the maximum height of the smoke plume (m);
QH is the heat release rate for a section of fire that contributes to plume rise (cal/s); and
U is the average wind speed during burn (in meters per second, m/s).

Wind speed and stability are parameters that vary during the day and year. To calculate plume rise, an average wind speed and stability during the burn will probably be used. For prescribed burns, the target meteorological conditions during which the burn will be conducted will be used to set these parameters. The parameters used for a wildfire will depend on the desired scenario (duration of burn, assumed weather conditions, etc.). It is also possible (but cumbersome) to vary these parameters on an hourly basis (which is the time step used in the meteorological data) using various options available in ISCST3 if a specific burn scenario with detailed resolution is desired.

Once plume rise has been calculated, the initial vertical dimension of the area source or sources that represent the fire can be determined. The *User's Guide for the Industrial Source Complex (ISC3) Dispersion Models, Volume I—User Instructions* (EPA, 1995b) recommends setting the initial vertical dimension to the plume height divided by 2.15 for surface-based sources, such as a fire.

To complete the discussion of modeling fire scenarios, it should be noted that any burn will only be modeled for the hours representing the burn duration and any desired time before or after the fire.

Appendix B

Table B-1. Soil Resuspension Radionuclide Emissions for Calendar Year 1998

Isopleth or Project[a]	Isotope Emissions (Ci/yr)[b]				
	Pu-239/240	Am-241	U-233/234	U-235	U-238
Isopleth 1	4.7E-07	7.6E-08	2.2E-08	5.3E-09	5.5E-10
Isopleth 2	1.6E-09	4.5E-09	--	7.9E-11	4.0E-09
Isopleth 3	3.1E-07	9.5E-07	--	1.7E-09	1.4E-07
Isopleth 4	2.4E-06	3.9E-07	--	7.1E-10	1.1E-09
Isopleth 5	--	2.1E-07	--	1.5E-09	--
Isopleth 6	1.0E-07	2.2E-07	--	7.2E-09	--
Isopleth 7	4.5E-09	2.6E-07	--	--	--
Isopleth 8	1.7E-06	2.1E-07	--	--	--
Isopleth 9	2.7E-06	4.7E-07	--	--	--
Isopleth 10	9.2E-07	6.5E-07	--	--	--
Isopleth 11	1.4E-06	1.1E-06	--	--	--
Isopleth 12	3.8E-06	--	--	--	--
Isopleth 13	5.9E-06	--	--	--	--
Isopleth 14	1.2E-10	--	--	--	--
Isopleth 15	2.9E-09	--	--	--	--
Isopleth 16	1.1E-05	--	--	--	--
Isopleth 17	5.8E-09	--	--	--	--
Isopleth 18	4.8E-07	--	--	--	--
Isopleth 19	5.3E-10	--	--	--	--
Isopleth 20	8.2E-10	--	--	--	--

[a] Isopleths are specific to each isotope and indicate zones of equal radionuclide emission potential for contaminated surface soils. Resuspension emissions calculated using Site-specific factor and 1998 meteorological data (DOE, 1999).

[b] Emissions of all isotopes that could contribute greater than 10% of the potential effective dose equivalent for a release point were estimated. Isotopes for which emissions were not estimated are shown as "--". The locations of the centroids of the referenced isopleths are shown in Figures B-1 through B-5.

Notes

Ci/yr = Curies per year
Pu = Plutonium
U = Uranium
Am = Americium
-- = Not estimated
E# = x $10^{\#}$

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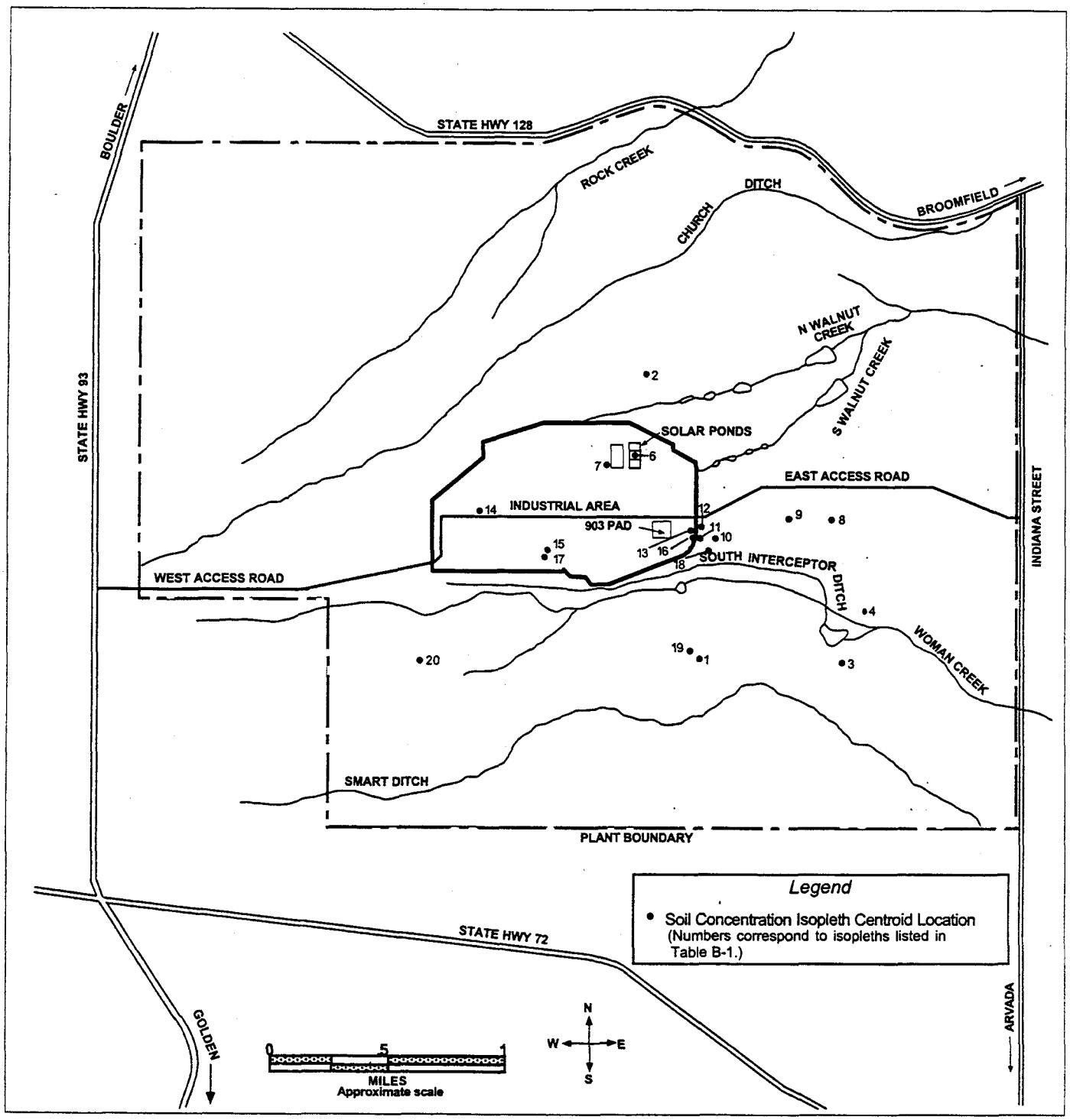

Figure B-1. Soil Concentration Isopleth Centroid Locations for Plutonium-239 and Plutonium-240

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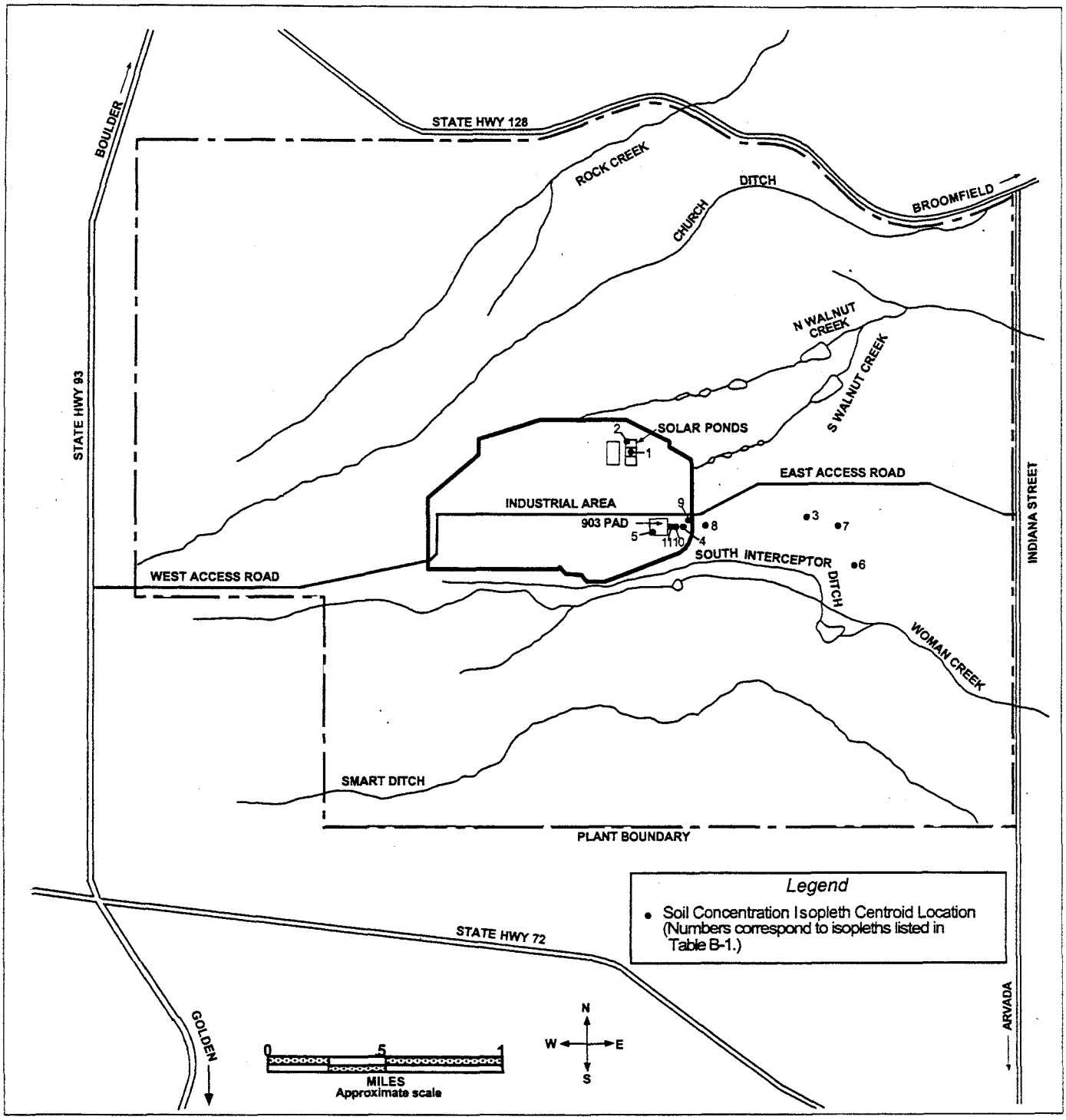

Figure B-2. Soil Concentration Isopleth Centroid Locations for Americium-241

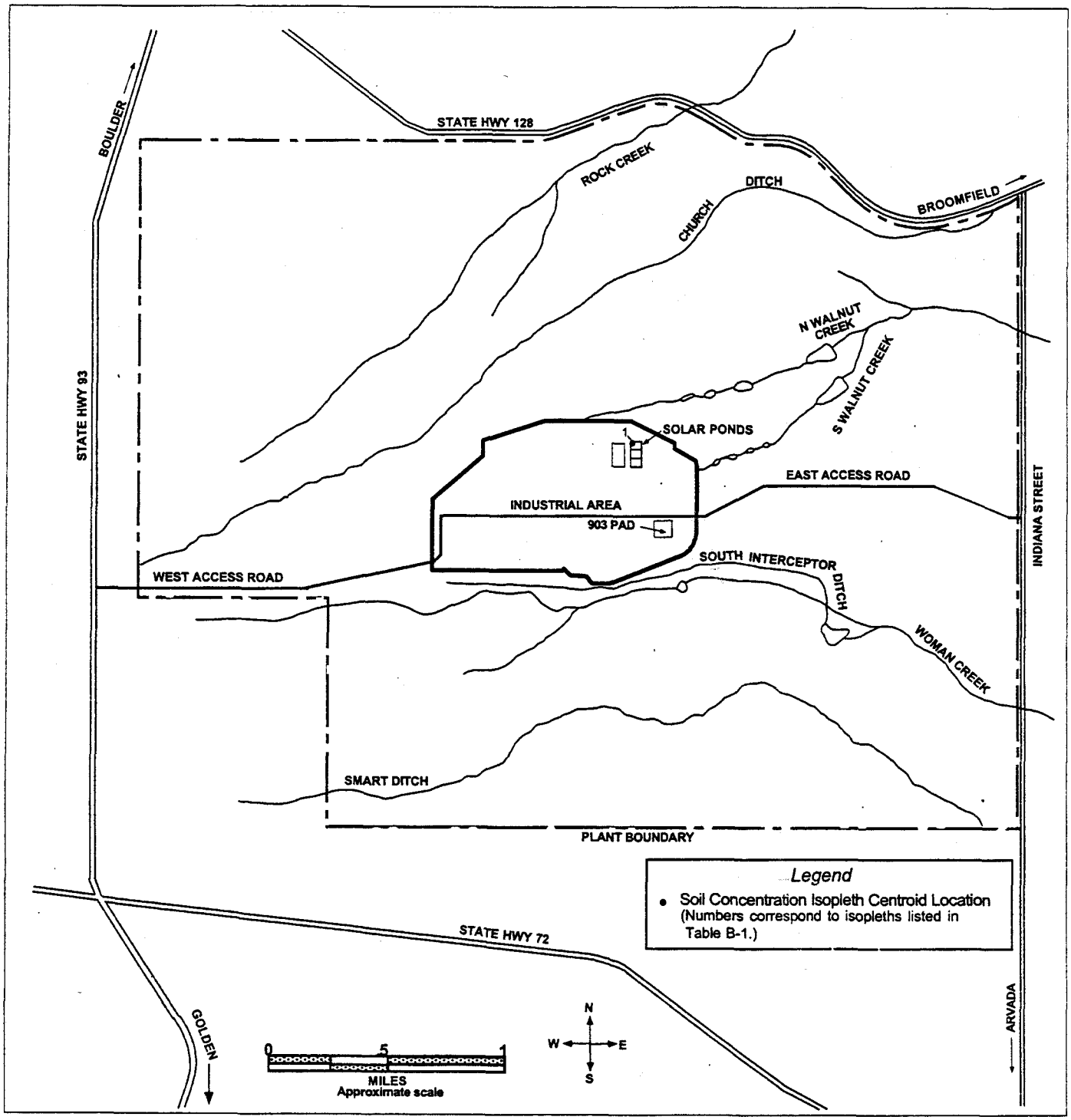

Figure B-3. Soil Concentration Isopleth Centroid Locations for Uranium-233 and Uranium-234

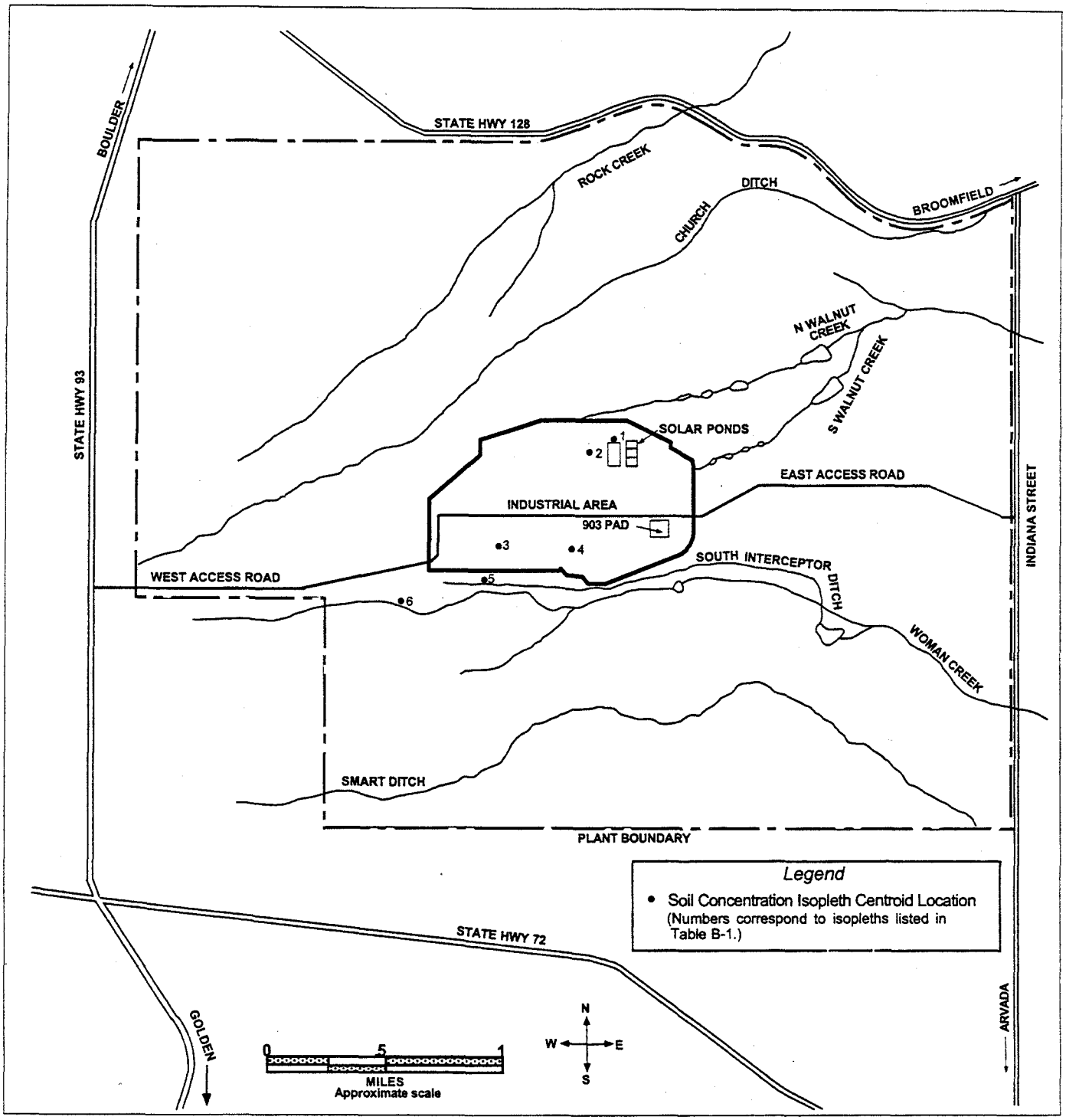

Figure B-4. Soil Concentration Isopleth Centroid Locations for Uranium-235

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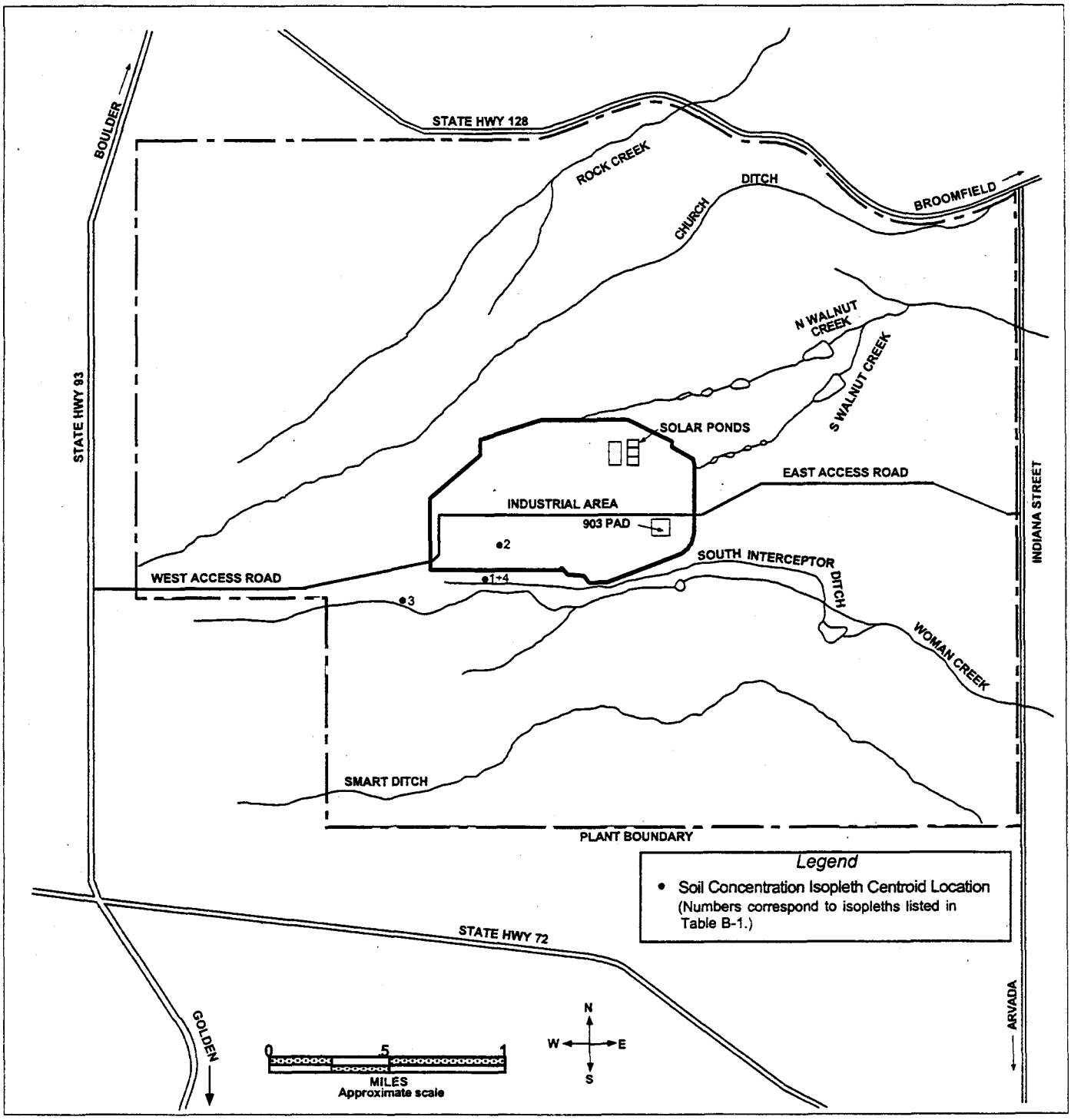

Figure B-5. Soil Concentration Isopleth Centroid Locations for Uranium-238

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Appendix C

Summary of Trend Analyses of Pu-239 Concentrations at S-107 Ambient Air Sampler

Purpose and Scope

During the months of January 1997, January 1998, February 1999, and March 1999, an order of magnitude increase in concentrations of plutonium-239 (Pu-239) and americium (AM-241) has been observed in samples taken at the ambient air sampler S-107, when compared to the remaining months of each year. S-107 is located at the northeast corner of the 903 Pad, a location that has historical actinide contamination.

An analysis of historical Pu-239 concentrations at the S-107 location has been conducted to determine if this trend was exhibited prior to 1997. Concentration data were also compared to historical meteorological data to try to identify a relationship between meteorological events and concentration increases. Am-241 concentrations were not used in the quantitative analysis because limited historical data are available prior to 1997.

Qualitative differences have been observed in Pu-239 concentrations between samplers S-007 and S-107, which were collocated and operated simultaneously from January 1995 to July 1998. However, statistical analyses of the paired S-007 and S-107 concentration data populations for Pu-239, using both a paired data sign test and the Kruskal-Wallis test, indicate that S-007 and S-107 data are not statistically different. S-107 concentration data are statistically different from S-007 concentration data only when monthly average wind speed exceeds 11.0 miles per hour, which occurs in only 18% of the paired data set. (The significant differences in concentration data at higher wind speeds is likely due to the higher-volume sampling rate of S-107, compared to S-007, which may allow it to extract more particulate matter from the airstream.) Therefore, S-107 data was used in lieu of S-007 data for the analyses reported here when both sets of data were available. S-007 ceased operation in July 1998; S-107 remains in service.

Observations and Hypotheses

Pu-239 Concentration Trends

Pu-239 concentration data at S-007/S-107 from October 1993 to April 1999 were available for this analysis. Pu-239 concentrations were recorded for either monthly or quarterly sampling intervals. Several periods of missing data exist during this time period, and differences exist in the intervals over which concentration and meteorological data were generated (quarterly versus monthly),

which limits the reliability of some data set comparisons. Qualitatively noteworthy increases in concentrations occurred during the winter months (December through March) relative to other months of the year in 1992, 1994, 1996, 1997, 1998, and 1999.

Hypotheses

The periodicity of concentration peaks suggests that the higher concentrations correlate with meteorological factors. A preliminary hypothesis for the increase in Pu-239 concentrations at S-107 was that a combination of high average wind speed and prevailing westerly wind direction during the winter season increased actinide-laden dust resuspension from the 903 Pad area, moving it eastward toward S-007/S-107. With this hypothesis in mind, meteorological and Pu-239 concentration data from October 1993 to April 1999 were analyzed and compared using the following scenarios:

Scenario	Function of Meteorological Factors
A	f(A) = (monthly average wind speed times the percentage of wind direction from the WNW, W, and WSW)
B	f(B) = [(monthly average wind speed times the percentage of wind direction from the WNW, W, and WSW) divided by total monthly precipitation]

Scenario A: f(A) = (monthly average wind speed times the percentage of wind direction from the WNW, W, and WSW)

This scenario tests whether a combination of relatively high monthly average wind speeds with a predominantly westerly wind direction correlate with elevated Pu-239. This relationship would suggest that stronger westerly winds blowing over the 903 Pad toward the S-007 and S-107 samplers may be a factor in elevated Pu-239 concentrations.

Scenario B: f(B) = [(monthly average wind speed times the percentage of wind direction from the WNW, W, and WSW) divided by total monthly precipitation]

This scenario tests whether a combination of relatively high monthly average wind speeds with a predominantly westerly wind direction **and** relatively low precipitation, as typically occurs from December to January, correlates to elevated Pu-239 concentrations in the air east of the 903 Pad. This relationship would suggest that stronger westerly wind blowing over dry ground may be a factor in elevated Pu-239 concentrations.

Results of Statistical Analyses

The Kruskal-Wallis test for independent data sets was used to determine whether statistically significant differences existed in the concentration of Pu-239 at S-007 and S-107 (S-107 data used when both data sets available) using the hypotheses proposed above. Correlation coefficients and t-statistics were developed and p-value tests performed to verify test outcomes.

A confidence level of 95% was used for these analyses ($\alpha = 0.05$). At $\alpha = 0.05$, the Chi-square quantile for the Kruskal-Wallis test is 3.84. When the Kruskal-Wallis test statistic is less than the Chi-square value, the null-hypothesis (no difference between populations) is accepted. When the Kruskal-Wallis test statistic is greater than the Chi-square value, the null-hypothesis is rejected and the data sets may be considered to be drawn from different populations.

In Scenario A, the total population of Pu-239 concentration data was subdivided into two groups, based on $f(A) \geq 400$ or $f(A) < 400$. The dividing value of $f(A) \geq 400$ was approximately 60% of the greatest f(A) value observed (674.2) and was arbitrarily selected based on a visual evaluation of a graph of Pu-239 concentration versus f(A).

In Scenario B, the total population of Pu-239 concentration data was subdivided into two groups, based on $f(B) \geq 1300$ or $f(B) < 1300$. The dividing value of $f(A) \geq 1300$ was approximately 20% of the greatest f(B) value observed (7194.0) and was arbitrarily selected based on a visual evaluation of a graph of Pu-239 concentration versus f(B).

Scenario	Results of Kruskal-Wallis Tests
A, all data[a]	KW statistic = 4.75, reject null-hypothesis (populations are different)
B, all data[a]	KW statistic = 0.07, accept null-hypothesis (populations are not different)
A, S-107[b]	KW statistic = 3.99, reject null-hypothesis (populations are different)
B, S-107[b]	KW statistic = 0.39, accept null-hypothesis (populations are not different)

[a]October 1993 – April 1999 (S-007 and S-107, S-107 data used when both available)
[b]January 1997 – April 1997 (S-107 only)

Because the meteorological factors used to subdivide the data set were arbitrarily selected, correlation coefficient (CC) tests were performed to confirm the Kruskal-Wallis test outcomes. A p-value of <0.05 was considered to be

significant and confirm that the observed correlation was not random chance (i.e., the probability that the observed correlation is random was less than 5%). Because the Kruskal-Wallis test uses ranked Pu-239 concentration data to test significance, p-values were determined for the data sets by correlating both rank to Scenarios A and B and actual Pu-239 concentration to Scenarios A and B.

The outcome of the p-value tests was affected by data ranking only in the case of f(A) for October 1993 – April 1999 (S-007/S-107). Whereas ranked data shows <1% likelihood of random correlation to westerly wind strength, the correlation of actual concentration to westerly wind strength shows about a 6% likelihood of being random. This may be due to differences in data collection intervals that occurred prior to January 1997, as discussed above. For the purpose of this summary, the p-value of 0.063 is not considered to detract significantly from the conclusion that actinide activity at S-007/S-107 has a strong correlation to westerly wind strength.

Scenario	Results of Correlation Coefficient Tests
Jan 1997 – Apr 1997 (S-107)	
A, by rank	CC = 0.4103, p-value = 0.003 (correlation <1% likely to be random)
B, by rank	CC = 0.0794, p-value = 0.694 (correlation likely to be random)
A, by conc.	CC = 0.5226, p-value = 0.005 (correlation <1% likely to be random)
B, by conc.	CC = 0.0198, p-value = 0.922 (correlation likely to be random)
Oct 1993 – Apr 1997 (S-007/S-107)	
A, by rank	CC = 0.2936, p-value = 0.031 (correlation ~3% likely to be random)
B, by rank	CC = 0.0542, p-value = 0.697 (correlation likely to be random)
A, by conc.	CC = 0.2548, p-value = 0.063 (correlation ~6% likely to be random)
B, by conc.	CC = 0.0555, p-value = 0.690 (correlation likely to be random)

Conclusions

Data from ambient air sampler S-007/S-107 show a trend of elevated Pu-239 concentrations during the winter season. Meteorological phenomena including strong, persistent westerly winds and low precipitation appeared to correspond to these Pu-239 concentration spikes, and suggested a causal relationship. Dust resuspension around the 903 Pad would likely be maximized under such conditions, so actinide concentrations in ambient air would be expected to increase.

Statistical tests of the available data, comparing Pu-239 concentrations to average monthly westerly wind strength and to average monthly westerly wind strength divided by total monthly precipitation, demonstrate that Pu-239 concentrations at S-007/S-107 strongly correlate to westerly wind strength but show little correlation with the inverse of monthly precipitation. Based on these statistical

tests, it may be predicted that actinide activity at S-107 may be expected to rise in proportion to westerly wind strength.

Only meteorological factors were considered in these analyses. Anthropogenic activities near the sampler locations could also cause increased actinide concentrations, although no significant activities were identified during the 1997 and 1998 periods that showed increased Pu-239 concentrations. Anthropogenic activities have not been examined in detail. Such activities could explain variations in concentration patterns, as seen from 1990 to 1995, when the graph of correspondence between westerly wind strength appears less causal.

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